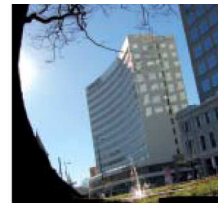
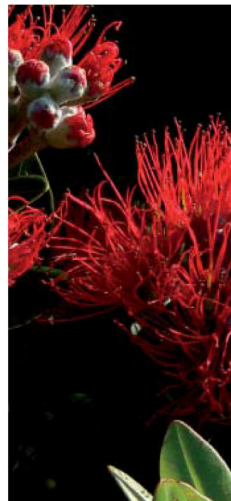


Carbon Stocks and Changes in New Zealand's Soils and Forests, and Implications of Post-2012 Accounting Options for Land-Based Emissions Offsets and Mitigation Opportunities - Including Appendices

Miko Kirschbaum, Craig Trotter, Steve Wakelin, Troy Baisden, Denis Curtin, John Dymond, Anwar Ghani, Haydon Jones, Markus Deurer, Greg Arnold, Peter Beets, Murray Davis, Carolyn Hedley, Duane Peltzer, Craig Ross, Louis Schipper, Anne Sutherland, Hailong Wang, Mike Beare, Brent Clothier, Norman Mason, Murray Ward



Landcare Research
Manaaki Whenua

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1. Goals and Methods

This report has been prepared for the Ministry for Agriculture and Forestry (MAF), as part of contract CC MAF POL_2008-12 (105-1). The overall goal of the contract was to:

Determine the implications for NZ of post-2012 accounting options on LULUCF-sector removals and mitigation potential, and identify and prioritise the knowledge gaps and uncertainties that most limit the reliability of the LULUCF component of forecasts of NZ's net emissions position.

In the work presented here we have:

- identified, critically reviewed, and documented the key characteristics and uncertainties in existing datasets that could contribute to analyses of the implications of post-2012 accounting options for LULUCF activities and land-based emissions mitigation potentials;
- analysed existing and newly-identified datasets to estimate the likely contribution of the LULUCF sector to New Zealand's net emissions position under post-2012 accounting options for the range of LULUCF activities expected to be accountable in the future (e.g., those in Articles 3.3 and 3.4);
- quantified where possible the sensitivity of the net emissions position forecasts under post-2012 accounting options to knowledge gaps and uncertainty in LULUCF data, and recommended a prioritised work programme to remove the most critical uncertainties.

2. Exotic Forest Sinks and Mitigation Options

The total exotic forest area is about 1.80 million hectares (Mha), with Radiata pine (*Pinus radiata*) comprising about 89% of the total area, and with Douglas-fir (*Pseudotsuga menziesii*) the next most common species at just 6%. About one-third of these forests have been established since 31 December 1989, and are thus “Kyoto forests”.

Existing estimates and forecast of carbon stocks in planted exotic forests are presently based on data from the National Exotic Forest Description (NEFD). In future, it is likely that estimates of planted forest carbon stocks will be made using a plot-based sampling approach in conjunction with LiDAR data, under the Ministry for the Environment's (MfE) LUCAS inventory programme. However, the relationship between these two sources of estimates remains uncertain.

The key strength of the NEFD-based approach is the wealth of age class data it contains. This allows forecasts to depict accurately the changing effects of forest age class, which have a very large impact on total carbon stocks and change. The primary weakness of the approach is that NEFD yield tables (of volume by age) may underestimate wood volume in post-1990 forests, which tend to be on more fertile sites, and so grow more rapidly—although such effects are likely to be compensated to a substantial degree by wood on more fertile sites having a lower wood density.

Using these data sources, it is estimated that planted exotic forests in New Zealand provide about 84 Mt CO₂ of removal units during the first commitment period (CP1) and will thereby make a substantial contribution to New Zealand's ability to meet its Kyoto targets over CP1. Deforestation, primarily from pre-1990 exotic forests, is currently estimated to contribute a liability of about 17 Mt CO₂ over CP1. Whether a liability of that magnitude will actually eventuate may depend largely on

the extent to which deforestation will be discouraged through policy means, such as the ETS, in future years. The contribution planted exotic forests can make as net carbon sinks over future commitment periods depends strongly on trends in future rates of both afforestation and deforestation as well as on future accounting rules.

Forecasts of carbon stocks and change in planted exotic forest show that under a gross-net accounting regime, with a 1990 baseline year for determining eligible forest land:

- At present planting and deforestation rates, post-1989 carbon stocks will continue to rise until about 2020 and then decline for about a decade, which means that these forests will become a net source of emissions during that time.
- A minimum planting rate of about 65 000 ha per year is required to prevent post-1989 exotic forests from becoming a net emissions source in any five-year commitment period (CP). With a planting rate of 20 000 ha per year, post-1989 forest stocks will still decline from about 2020, and by about 2030 will reach a minimum of about 70% of 2020 levels. Carbon stocks will increase again after that.

It is likely that New Zealand could gain significant additional forest sink credits during CP1 from a downward revision of the estimated percentage of over-planting of indigenous shrubland (i.e. exotic forests planted on land that previously contained forests so that it would not be classed as land-use change and could not generate eligible removal units), and of soil carbon losses associated with afforestation of grassland—particularly in eroded landscapes. Progress in improving data quality in these areas has been very slow.

Any move to all-forests net-net accounting with a 1990 baseline year, or a multi-year baseline centred on 1990, will be strongly disadvantageous to New Zealand. Pre-1990 forest removals peaked around that time, and all-forest removals peaked about 2006. Unless a very substantial expansion in future planting occurs, planted forests in New Zealand would therefore be accounted as a net emissions source beyond CP1 under most circumstances. By way of comparison, accounting during CP2 (taken as 2013–2018) is forecast to result in net accountable removals by forests of about 77 Mt CO₂ under the same accounting regime as applied during CP1, but net accountable emissions of about 10 Mt CO₂ under an all-forests net-net accounting regime with a 1990 baseline. Forecasts over a longer term, out to 2050, show continuing substantial disadvantage to New Zealand under an all-forests net-net accounting regime, even under scenarios involving average new planting rates of 20 000 ha per year.

The following forest management options could be considered to increase forest carbon sinks under the current gross-net or any form of net-net accounting rules:

- Increasing rotation age.
- Use of alternative species, especially those suited to longer rotations.
- Increasing volume and/or wood density through genotype selection and/or modifying stocking.

Retaining the pre/post-1990 split for planted exotic forests, and accounting carbon stocks in pre-1990 forests on a net-net basis—with post-1990 forests accounted on a gross-net basis—may be an option New Zealand wishes to further consider. Such a split-accounting approach appears to be substantially more favourable than an all-forests or net-net accounting approach—though still

considerably less favourable than the CP1 gross-net (post-1990) plus deforestation-liability (pre-1990), approach. More work would need to be done on a split-accounting option to determine precisely the implications for New Zealand if the option were to become a serious international contender, as it was not considered in detail in this study.

For a split-accounting approach to have the most favourable outcome for New Zealand, incentives would need to be introduced to:

- minimise liabilities from deforestation of pre-1990 forests (including through advancing the concept internationally of “land-use flexibility mechanisms”;¹
- minimise liabilities from deforestation of post-1990 forests (including through extension of the “fast forest fix”² to cover both harvesting and deforestation of post-1990 forests);
- adopt forest management options for forests subject to net–net accounting as outlined above—extending rotation time in particular. International adoption of New Zealand’s proposed “land-use flexibility” mechanism for pre-1990 forests would greatly assist in ensuring pre-1990 forests accrued, at the very least, no liabilities;
- annually plant at least 20 000 ha of new forest onto grassland to avoid post-1990 forests becoming a net source at some point in the future.

Various analyses have shown that there are at least 1 Mha of marginal, severely erosion-prone, pastoral lands that would benefit from exotic afforestation—and potentially as much as 2.6 Mha if moderately erosion-prone lands are included as well. This provides sufficient scope to support a planting programme, with associated environmental co-benefits, of at least 20 000 ha per year until 2050. Carbon accumulation on such lands is expected to average about 25 t CO₂ per ha per year.

The marginal lands are in the following ownership classes:

- Crown-owned land: 0.13 Mha (or 0.25 Mha³).
- Privately-owned land: 0.92 Mha (or 2.36 Mha).
- Maori-owned land included in privately-owned land: 0.05 Mha (or 0.11 Mha).

The major knowledge gaps and uncertainties that most limit the reliability of estimates and forecasts of the contribution planted exotic forest net removals make to New Zealand’s net emissions position are (in order of priority):

- The most likely post-2012 accounting regime—this has by far the largest impact on the modelling and forecasting of forest emissions/removals, with all-forests or net–net approaches being very disadvantageous to New Zealand compared with CP1 accounting rules.

¹ The land-use flexibility option being advanced by NZ allows for an area of pre-1990 exotic plantation forest to be harvested, and replanted in another location, without being considered deforestation.

² The “fast-forest fix” states that debits resulting from harvesting during the first commitment period following afforestation and reforestation since 1990 shall not be greater than credits accounted for on that unit of land.

³ The alternative, larger areas are those that result if lands with a moderate erosion risk are included.

- Limitations of the NEFD-based approach to modelling in terms of applicability to post-1989 forests (effect of soil fertility and unknown management regimes), and inability to provide estimates with statistically based confidence limits. Inventories completed under MfE's LUCAS approach may address some of these issues. There are also calibration/validation issues to be addressed with the proposed LUCAS approach itself, but these are beyond the scope of this report.
- An urgent requirement is for available LUCAS data for planted forests to be analysed and used to calibrate or replace both the post-1989 and pre-1990 NEFD-based yield tables. Verification of NEFD and other planted forest areas from LUCAS mapping work, if sufficiently reliable, is also a priority.
- Improved characterisation of the rates and forecasts of deforestation and harvesting. This includes effects on the soil carbon pool, especially if areas of land-use change become larger under (currently proposed) land-use flexibility mechanisms. It also includes development of least-cost remote sensing techniques (probably based on satellite radar imagery) to map deforestation and harvesting.
- Development of carbon partitioning and wood density functions for Douglas fir, and to a lesser degree for eucalypts.
- Development of reliable defaults for understorey carbon stock changes following planted exotic afforestation, and for emissions from pre-afforestation land clearance, as a function of climate/soil conditions. Net-net accounting may also require estimation of pre-afforestation sequestration rates, depending on the form of the baseline approach.
- Validation of assumptions in relation to the proportion of carbon stocks removed off-site and residues remaining, particularly if residues become a source material for biochar or bioenergy.
- Development and validation of indices related to forest management activities in planted forests that can be determined using remote sensing techniques.

3. Indigenous Forest Sinks and Mitigation Options

Changes in carbon stocks of indigenous forest in New Zealand are not presently included in the accounting of emissions or removals under the Kyoto Protocol unless these forests are involved in a land-use change. The inclusion of carbon-stock changes resulting from forest management was voluntary for CP1, and New Zealand chose not to include them. However, they are required to be reported under the UNFCCC. It is currently assumed that these are old-growth forests that are carbon-neutral, although this is supported by very little quantitative evidence.

In future commitment periods, emissions or removals by these forests may have to be accounted under some of the variants of all-forests or net-net accounting approaches. Because the area of indigenous forests is relatively large (c. 6 Mha), carbon-stock changes of even a few tonnes per hectare per year could have significant implications for New Zealand's carbon balance. Deforestation of indigenous forests is, however, minimal, and will result in little liability in future commitment periods.

Work completed in this study analysed a substantial number of inventory plots for changes in the live biomass pool. It used 206 plots which is a much larger set than the 39 plots that have been used

in previous work. The analysis of the indigenous forest live carbon pools found no statistically significant carbon-stock changes over time.

The results for the dead wood pool are the first reported, and are based on time-sequence studies on a set of 31 inventory sample plots that have at least 4 sets of measurements. The analysis indicated that it takes 30 years on average for the coarse woody debris (CWD) pool to be reduced by 50% (the decay half-life). The value was determined from studies using 7 indigenous tree species, of which 5 were among the 10 most abundant species in New Zealand. Decay rates differed significantly between some of these 7 species. However, using a mean decay rate was nonetheless considered to be appropriate given that the species identity of CWD is often difficult to determine and is therefore often recorded in plot inventory data as being unknown.

Indigenous forestation of marginal lands offers considerable potential for both emissions mitigation, with co-benefits of erosion control, shifting to a more sustainable land use and increasing indigenous biodiversity. Depending on the potential erosion severity rating that is used to define lands as “marginal”, between 4.6 Mha and 2.7 Mha of marginal pasture lands are available, with about 60%, or 40%, respectively, of these lands in private ownership. Using indigenous forests only, forestation of all marginal lands would result in carbon sequestration over the active growth phase (lasting at least 150 years) of 24 Mt CO₂ yr⁻¹ or 14 Mt CO₂ yr⁻¹, respectively, for the two classes of marginal lands.

There are no major limitations, critical assumptions, large uncertainties or substantial knowledge gaps involved in making estimates and forecasts of existing indigenous forest carbon stocks. Although future cycles of LUCAS inventories will provide valuable confirmatory information, analysis of data available to date has shown no significant change in live biomass stocks over time. Moreover, although our knowledge of carbon stocks in the dead-wood pools remains preliminary, it is unlikely that these are changing significantly if live biomass stocks are also not changing. This is confirmed for such quantitative analysis as is possible to date (on just 31 inventory plots), although clearly a larger study is yet required before this can be fully confirmed.

The knowledge gaps and uncertainties that most limit the reliability of the estimate of carbon sequestration in re-established indigenous forests to New Zealand’s post-2012 net emissions position, and mitigation options, are:

- the need for a more precise definition of marginal pasture land, and of the carbon price at which “carbon farming” on such land becomes a viable economic proposition—by region, and probably by land classes within regions;
- a lack of models of regional, and preferably sub-regional, rates of carbon sequestration in indigenous forests based on likely successional pathways;
- little current effort in developing and validating models of indigenous forest establishment, including of rates of canopy closure under natural regeneration regimes—and for establishment and growth of indigenous shrublands;
- a lack of information on land management practices that can enhance natural regeneration rates for indigenous forest, and that can encourage rapid succession from lower-biomass shrubland to higher-biomass tall forest.

4. Soil Carbon Sinks, Sources and Mitigation Options

4.1 Afforestation/ Reforestation/ Deforestation

For the First Commitment Period, only soil carbon changes related to land use change (under Article 3.3 of the Kyoto Protocol) need to be accounted because New Zealand chose not to include any of the optional components under Article 3.4. In New Zealand, that principally means soils carbon changes after deforestation, the conversion of established forest to pastures or other land uses, and afforestation or reforestation, the conversion of pastures to native or exotic forests. Soil carbon is generally lost following afforestation/ reforestation with exotic forests which is an important issue for New Zealand as it reduces the benefit of biomass carbon stocks in newly planted forests, and it affects large areas of the country.

Different analyses have derived estimates of soil carbon losses between 8 and 18 t C ha⁻¹ on conversion of pasture to pine forests. In particular, the wide soil sampling that had been incorporated in New Zealand's Carbon Monitoring System had developed a best estimate of a loss of soil C of 18 t C ha⁻¹, whereas analysis of paired sites had derived a lower estimate of 8.5 t C ha⁻¹. A newer analysis reported as part of this review has explicitly taken the auto-correlation between different sampling points into account, and has derived lower estimates of between 8 and 13 t C ha⁻¹. The various analyses all carry large uncertainties so that the different estimates are not statistically different from each other. Given some reasonable assumptions about the costs of carbon and the magnitude of the areas that have been reforested in New Zealand, the difference between these estimates translates into differences in the size of New Zealand's greenhouse gas liability of the order of \$150M. Additional work to further refine that estimate is thus considered to be a high priority.

As important as the magnitude of change is its time course. The current default for accounting purposes is a linear change over 20 years. It is possible, however, that the change is not linear but loss may be more rapid over early years after land-use change and then stabilise towards a new value, with lesser changes in later years. Such a time course would be advantageous for New Zealand as rates of plantation establishment were particularly high in the early 1990s. Under the current IPCC default, any changes in soil carbon calculated based on a linear-change assumption have to be carried through the first commitment period, whereas the liability would be substantially less if the time course is sigmoidal or exponential, and most of the change has already occurred.

Measurements of the time course of soil carbon losses are unlikely to be detailed enough to be able to provide definitive answers on the likely time course of change, and the small number of modelling studies that have addressed the question have not yet given a clear and unambiguous answer to that question. This, too is an area where further work could be most useful in reducing New Zealand's accounting liability.

For the deforestation of exotic forests to pasture, it is presently assumed that the soil C loss incurred in a shift from pasture to forest would simply be reversed. There are very limited data to support this assumption, and the only recent New Zealand study of soil C changes upon deforestation has observed very large gains in soil C. With the recent upsurge of conversions of forests to dairying, it is warranted to expend some further efforts on quantifying the soil C change following deforestation.

Soil sampling to date has also purposefully excluded sites affected by recent erosion events, but since erosion is part of the experience at different sites, locations affected by erosion should, in fact, be included in any representative and carefully stratified sampling regime. It is therefore not known

if the land-use factors currently used are appropriate for erodible hill-country. Carbon accounting with erosion raises other important issues that are discussed further below.

Regardless of the post-2012 accounting regime that will finally be endorsed internationally, changes in carbon stocks with afforestation/reforestation will have to be quantified and accounted. Under gross-net accounting approaches, as is currently done under Article 3.3, soil C changes partly negate the larger and opposite change in biomass C stocks. So, soil C changes reduce the benefit from reforestation, and reduce the liability from deforestation.

If net-net accounting were to be adopted post 2012, the loss of soil C with reforestation would actually be advantageous as any reduced C gain would increase total 1990 baseline emissions whereas future soil-C losses would trend towards zero as more and more forests reach an age beyond the assumed 20 years for adjustment in soil C stocks. Net-net accounting in any form would carry many other major disadvantages for New Zealand, however, so the issue of soil carbon change would be of relatively minor importance.

4.2 Biochar Amendment

The use of biochar has been proposed as a means to store more carbon in soils to improve the net greenhouse gas balance of various agricultural or forestry practices. However, to assess the full greenhouse implications of biochar addition, it is necessary to consider a range of processes and interactions at different time scales. It is, therefore, not possible to calculate the greenhouse benefit of biochar addition by simply adding the amount of biochar carbon stored to the greenhouse balance that would be obtained without the addition of biochar. In this report, we provide a mathematical framework for estimating the net carbon balance associated with the addition of biochar to cropland or pastoral soils.

Calculations performed using the framework broadly confirm that there is a substantial potential for C sequestration benefits from biochar incorporation in New Zealand soils. Nevertheless, the calculations also suggest considerable uncertainty stemming from uncertainty around some key parameters, with pessimistic calculations suggesting that over 5 years, biochar application to croplands might barely result in any net C storage, and might even be a net C source to the atmosphere for application to pastures when incorporation requires tillage. There is a wide range of values between estimates based on pessimistic and optimistic parameter settings, but in all cases it is clear that the net carbon sequestration resulting from biochar addition to soils will be considerably less than the quantity of added biochar.

This work shows the importance of introducing robust equations for accounting for biochar addition under New Zealand conditions so that major uncertainties can be identified and targeted for future research. The main uncertainties relate to the accounting of the diverted biomass used to produce biochar, the residence times of soil C and the dynamic fraction of biochar, the proportion of biochar that is effectively resistant to decomposition, and the loss of soil C resulting from tillage and biochar incorporation.

4.3 Forest Management

Soil C changes that result from forest management need not be accounted for during the First Commitment period of the Kyoto Protocol. Forest management is included under Article 3.4 which may become mandatory for future Commitment Periods depending on the outcome of international negotiations. Soil C under forests has been shown to change under silvicultural management, with carbon stocks generally increasing with tree stocking rates (up to 200 stems ha⁻¹), fertiliser application and retention of a weedy cover between rotations. On the other hand, carbon stocks

generally decrease with harvesting or site-preparation techniques that physically disturb the soil, or with complete removal of harvest residues and forest floor materials.

However, available data are insufficient to quantify the effects under New Zealand's conditions. Most available studies have not been conducted with carbon accounting as their main objective, and soil carbon measurements have either been collected for shallow depths, without bulk-density measurements, or under very specific soil and climatic conditions that make national extrapolations difficult. It is also not well known to what spatial extents different silvicultural practices are used throughout New Zealand.

In terms of post-2012 accounting, it is possible that the accounting of forest management will become mandatory under either Article 3.4, or under any form of net-net accounting. However, until further work is undertaken to better establish the full effects of forest management activities on soil carbon stocks, taking into account the areas of land affected at the national level, it is very difficult to be certain about the implications that effects of practices such as spot-mounding and ground-based harvesting might have on New Zealand's net position. Mitigation opportunities, such as retention of a weedy ground cover between rotations, are also limited due to their potential to interfere with normal site management for optimum wood production. Nevertheless, there are some practicable and well-established forest management practices available to forest managers that may help maintain or even increase soil carbon stocks (e.g., full residue retention on site).

There is some potential to include biochar in future operations, especially if small, mobile units can be developed that can make use of available harvest residue for combined bio-energy/biochar production. However, much more work needs to be done to assess whether the use of harvest residues for the production and application of biochar is indeed the most beneficial strategy in greenhouse gas terms, and whether biochar application has useful co-benefits or, instead, lead to some detrimental side-effects.

4.4 Pastoral Agriculture

Pastoral agriculture is New Zealand's dominant land-based activity, but carbon-stock changes do not need to be accounted for during the Kyoto First Commitment Period because New Zealand chose not to include grazing-land management, which is an optional component under Article 3.4 for the First Commitment Period. It has also long been assumed that soil carbon stocks would be highest under pastoral land use and that they would remain constant in the absence of any land-use change. Recent work, however, has suggested there may be changes within the broad classification of pastoral agriculture.

In particular, for dairy pastures on lowland non-allophanic soils, soil carbon stocks appear to have declined over the last 20 years. By contrast, on dry stock, hill country pastures, it appears soil carbon stocks have increased over the last 20 years. So far, no mechanism has been identified for these changes, and there is consequently little confidence in the magnitude of past trends, or for the prediction of future trends. It is also not clear why there appear to be different trends in different landscapes or production systems. If there are indeed differences, it is thus not clear whether those differences are related to terrain, soil type or production system, such as management intensity, fertiliser applications rates, and the degree of soil stability or disturbance.

It is also not yet known whether these losses and gains have occurred recently, have been occurring gradually over time, and are on-going, and whether they occur uniformly across the country. There is also little understanding of the processes controlling, and factors contributing to, such losses. At this stage, it still seems likely that the pastoral sector as a whole is neither gaining nor losing carbon, but further work is clearly warranted to better establish whether different parts of the

country may, in fact, display divergent trends, or more generally to better understand the factors that can lead to carbon gains or losses in pastoral soils. There is also a dearth of long-term monitoring of soil carbon below 7.5 cm depth. Currently available information is therefore of limited use both for determining management effects on total soil C stocks, and for meeting international requirements for estimating carbon stocks to a minimum depth of 0–30 cm.

Changes in soil C under pastoral soils may have to be accounted in post-2012 agreements under either Article 3.4 or under any form of net–net accounting. That is of little consequence if there is, indeed, no change in soil carbon. If there are identified changes, however, they could easily become important for national totals because of the large tracts of land involved.

There is little identified mitigation potential as standard pastoral management already leads to high soil C levels. There are risks, however, due to the emerging trend of soil cultivation as part of forage cropping to support pastoral agricultural systems. Cut-and-carry systems, which have recently begun to be considered for high-end production systems, have also not yet been assessed in terms of their C-stock implications. High-country tussock grasslands are also more vulnerable than lower elevation, more productive pastoral lands, and reductions in soil carbon have been observed in association with various forms of degradation of these lands, especially in relation to frequent burning and nutrient losses.

4.5 Cropping

Cropping on land that has remained under cropping since 1990 does not need to be accounted under the Kyoto Protocol because New Zealand chose not to include cropland management, which is an optional component for the First Commitment Period. It is also assumed that carbon stocks on these lands have stabilised at a new level by now. Cropland area is increasing by about 500 ha per year, and upon conversion, it is likely to lose about $0.5 \text{ t C ha}^{-1} \text{ yr}^{-1}$, for about 20 years. Provided conversion rates continue at these relatively small amounts, carbon losses can be expected to be about $0.02 \text{ Mt CO}_2 \text{ yr}^{-1}$, and thus make only a small contribution to the national total.

Mitigation opportunity consist of a reduction in soil disturbance (zero or minimum tillage) that reduces the rate at which organic matter in the soil breaks down, and maximum residue input and incorporation that principally involves a cessation of residue burning and retaining it on site. The mitigation potential of these options has not yet been satisfactorily quantified, mainly because most past work had not been conducted with the aim of carbon accounting and data have been collected at too shallow a depth, or without bulk-density measurements. A shift towards carbon mitigation practices might also entail other management difficulties that render theoretical options unsuitable in practice.

4.6 Horticulture

Like cropping, horticulture is gradually expanding, but by only about 1000–2000 ha per year. On average, horticultural soils are estimated to lose $9 \pm 7 \text{ t C ha}^{-1}$ on conversion from pastoral land, but that loss must be balanced by likely increases in biomass of a comparable magnitude. The combined carbon change is therefore likely to be very small and possibly even positive.

Horticultural operations generally do not aim to maximise biological productivity and often keep bare soil underneath their economic plants, both of which reduce the potential for organic C build-up. A shift to organic farming methods, or any practice that increases the input of residue carbon to the soil, is likely to increase the amount of soil C. There may also be opportunities for biochar incorporation, but the potential of any of these options has not yet been satisfactorily quantified, and questions remain as to their compatibility with standard management operations.

Whether C stock changes to or from horticulture need to be accounted depend on the type of horticultural crop being grown and whether it meets the definition of a forest (and depending on the previous land cover). The expansion of horticulture comes largely at the expense of pastoral land, and since combined carbon-stock changes in those conversions are likely to be small it does not matter greatly whether horticultural expansion is included in the accounting or not.

4.7 Erosion

Erosion raises complex questions both in terms of the overall carbon cycle and in terms of carbon accounting. In terms of the carbon cycle, the question is essentially whether erosion acts as an overall source or sink in terms of carbon fluxes to and from the atmosphere. In terms of carbon accounting, the question is how this atmospheric impact can be captured in accounting rules, or to what extent current accounting rules are inconsistent with the wider role of erosion as a component of the global carbon cycle.

Erosion in the first instance is simply the movement of carbon from one part of the landscape to another, with no immediate exchange with the atmosphere. In the longer term, net carbon emissions to the atmosphere can be increased if carbon input into the system is reduced through reduced biological activity on the erosion scars or if the eroded carbon is rendered more decomposable through the movement from its original location. Net carbon emissions can decrease if the displaced carbon becomes more resistant to degradation, which may occur if it is deposited in anaerobic deposits like lakes or ocean sediments. Decomposition may also be slowed if it is simply buried by other soil material, but the extent of this process is less certain. Overall, erosion and deposition become a carbon sink if decomposition of carbon in depositional sites is relatively slow, and if soil carbon is replaced on the eroded site by inputs to the soil carbon pool from vigorous plant production that returns relatively rapidly to pre-erosion rates.

We report here a first quantification of the likely effect of erosion on carbon fluxes from New Zealand's soils. This has included a detailed analysis of the types of erosion, their sediment yield, the associated carbon concentrations, and an assessment of the likely places where eroded carbon might be deposited. It also includes assumptions about the rate of recovery of soil carbon on erosion scars, the rate of carbon loss from deposits, and the proportion of material oxidised after deposition in the oceans.

The analysis suggests that overall, erosion in New Zealand constitutes a net sink of 0.85 Mt C/yr for the North Island, and a net sink of 2.3 Mt C/yr for the South Island, for a total of 3.15 Mt C/yr for New Zealand. For the South Island, the analysis indicates a river transport to the oceans of about 2.9 Mt C/yr, with about 0.6 Mt C/yr being oxidised in the ocean and new sequestration on land of 2.9 Mt C/yr, for a net flux out of the atmosphere of 2.3 Mt C/yr. For the North Island, the analysis indicates a river transport of about 1.9 Mt C/yr, sequestration of 1.25 Mt C/yr and oxidation in the ocean of about 0.4 Mt C/yr. This adds to a net flux from the atmosphere of 0.85 Mt C/yr, but also a loss of the amount of carbon stored on land by about 0.65 Mt C/yr. It is possible to have a net flux out of the atmosphere while land stocks are decreasing through an increase in the amount of carbon stored in ocean deposits.

While this analysis has attempted to quantify each of the relevant terms as carefully as possible, it must nonetheless be recognised that there is a dearth of data on some of the key parameters, such as the rate of recovery of soil carbon stocks on erosion scars and the rate of decomposition of eroded carbon both where it is deposited on land and in water ways. Because of the quantitative importance of this process, it would be warranted to expend further resources on a better quantification of the key rates and processes.

In terms of carbon accounting, erosion is generally counted as a source of soil carbon to the atmosphere. Accounting considers carbon stocks per unit of land, and if carbon is removed from a unit of land, it is considered as a soil carbon loss even if the carbon is simply transferred to a different pool, such as ocean deposits, without actually being lost to the atmosphere. Some difficulty will be encountered in attempts to modify the current accounting philosophy to adequately capture the net impact of erosion and deposition on the atmosphere, as well as on each unit that exists within the current accounting system. Therefore, the inclusion of projections for the erosion and deposition of carbon in national accounts is not presently recommended. It would nonetheless be warranted to develop accounting procedures for erosion and deposition in order to capture and properly account for the effects of this important process in an unbiased manner. Correcting carbon accounting frameworks to fully include erosion and deposition processes will take some years to develop, but may have net benefits to New Zealand.

5. Analysis of Different Accounting Options

We have evaluated the contribution of the LULUCF sector to New Zealand's net position under the major options being proposed for post-2012 accounting. Six different accounting options were considered:

- A status quo (Gross-Net) approach to LULUCF accounting.
- An All-Lands Gross-Net Accounting approach.
- An All-Lands Gross-Net Accounting approach with an additional cap on credits/debits from forest management.
- An all-lands net-net accounting approach referenced to 1990.
- A forward-looking baseline approach.
- The Average Carbon Stocks (ACS) approach.

Analysis of the LULUCF Sector of New Zealand's budget over the First Commitment Period clearly indicates that accountable net emissions are dominated by carbon uptake by exotic forests and carbon losses from deforestation. Carbon stock changes associated with cropping, pastoral agriculture, horticulture and forest management presently make quantitatively small contributions to New Zealand's overall reported position, either because net stock changes per units area are small or because the areas involved are small. The possible exception to this is pastoral agriculture: present results suggest gains in some landscape/ production systems and losses in others, but uncertainty remains high, and small overall net gains or losses are possible. As pastoral agriculture covers such a large area of New Zealand, even relatively small changes per unit area could add to a significant amount for the country as a whole.

For the Second, Third and further Commitment Periods, New Zealand's LULUCF emissions are expected to continue to be dominated by exotic plantations, deforestation and, provided that sufficient policy incentives are provided, the reestablishment of indigenous forests especially on marginal pastoral land. Whether deforestation rates will remain as high as in the recent past will depend on future trends in economic drivers, such as wood and dairy prices, and the extent to which disincentives, such as through the ETS, are maintained and enforced over future Commitment Periods. It is also again necessary to be cautious with respect to possible changes in soil carbon under pastoral agriculture, as any possible changes are yet to be adequately quantified.

Across the range of plausible mitigation options and accounting schemes, the Sector as a whole could potentially contribute large credits or debits. Plausible ranges were calculated as -120 to

+125 Mt CO₂ over the Second Commitment Period, and between –135 and 120 Mt CO₂ over the Third Commitment Period. This range is about equally due to possibilities for the success of mitigation options, such as the large potential for forest establishment and preventing deforestation, as to the range of possible accounting options.

As scenarios, it was assessed that future establishment rates of exotic forests might range between 0 and 40 000 ha per year, establishment of indigenous forests could potentially range between 0 and 100 000 ha per year, and deforestation rates could vary between 0 and 10 000 ha per year. Together, these options could have a combined mitigation potential of 90 Mt CO₂ over the Second Commitment Period and 140 000 Mt CO₂ over the Third Commitment Period. In terms of possible accounting options, continuation of the status quo (Gross-Net accounting), ‘All-Land Gross-Net Accounting with a Cap on Forest Management Emissions’, and application of the ‘Forward-Looking Baseline’ approach would lead to similar outcomes for the LULUCF Sector over the Second Commitment Period, and would be likely to generate credits similar to those anticipated for the First Commitment Period. These credits would diminish for the Third Commitment Period as the existing post-1989 estate reaches maturity and no longer generates further credits. On-going credits could only be maintained through substantial new plantings.

New Zealand’s net position would, however, be much worse under the ‘All-Land Gross Net Account’ (without a cap) because pre-1990 exotic forests would need to be included and these forests are anticipated to constitute a significant source over the Second and Third Commitment Periods. The worst possible outcomes for New Zealand would occur under application of ‘All-Lands Net–Net Accounting because New Zealand’s forests were a large sink in 1990 (by 95 Mt CO₂ per Commitment Period). If that uptake had to be included in the baseline it would worsen New Zealand’s net position by those 95 Mt CO₂.

Application of the Average Carbon Stocks approach would lead to lower credits for the Second Commitment Period than the most beneficial options. That differences between the Average Carbon Stocks approach and the other options becomes small by the Third Commitment Period as the post-1989 estate matures.

Overall, the range of possibilities due to uncertainties in the success of future mitigation policies, and about possible accounting options, far outweighs the scientific uncertainty about specific processes or extent of specific activities (with the possible exception of carbon-stock changes under pastoral agriculture). The outstanding priorities are therefore to:

- develop better scenario assessment tools to quantify the consequence of different accounting options and assess their implications for New Zealand’s net position;
- more completely quantify changes in soil carbon stocks under pastoral agricultural management regimes and soil types (including for forage cropping under low/no-till options);
- better determine the effects of cropping on soil carbon stocks, as cropland has become a key category for New Zealand.

CHAPTER TWO INTRODUCTION, GOALS AND METHODS

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1. Introduction

In the Kyoto Protocol's first commitment period, ending 2012, New Zealand's net emissions position will benefit substantially from accounting under the LULUCF sector of CO₂ removals by post-1990 exotic forests. However, the contribution the LULUCF sector makes to New Zealand's net position in future commitment periods will depend significantly on accounting rules yet to be finalised. It is likely that after 2012, accounting will become mandatory for a wider range of LULUCF activities (e.g., for those presently optional under the Protocol's Article 3.4). Accounting may also move to a fuller land-based accounting approach as compared with the current activities-based approach and is likely to become more rigorous.

Although the details of post-2012 accounting rules have yet to be decided, the major options are already known. Given the importance to New Zealand of emissions offsets and mitigation opportunities in the LULUCF sector—in both forest biomass and soils—it is imperative for the implications of post-2012 accounting options to be clear when developing either domestic climate change policy or international negotiating positions. It is also important to know the level of certainty with which conclusions on accounting implications can be drawn. At present, however, there are still significant gaps in the information required to determine realistically the effect a wider range of accountable LULUCF activities, and different accounting options, will have on New Zealand's post-2012 net emissions position. There has also not yet been a full identification of all sources of the best available New Zealand data that could be used to fill the remaining knowledge gaps.

In the work presented here we have:

- identified, critically reviewed, and documented the key characteristics and uncertainties in existing datasets that could contribute new information to analyses of the implications of post-2012 accounting options for LULUCF activities and land-based emissions mitigation potentials;
- analysed existing, and newly identified, datasets to estimate the likely contribution of the LULUCF sector to New Zealand's net emissions position under post-2012 accounting options, for the range of LULUCF activities expected to be accountable in the future (e.g., those in Articles 3.3 and 3.4);
- where possible, quantified the sensitivity of New Zealand's net emissions position forecasts under post-2012 accounting options to knowledge gaps and uncertainties in LULUCF data, and recommended a prioritised work programme to remove the most critical uncertainties.

2. Goal

This report has been prepared for the Ministry for Agriculture and Forestry (MAF), as part of contract CC MAF POL_2008-12 (105-1). The overall goal of the contract was to:

Determine the implications for New Zealand of post-2012 accounting options on LULUCF-sector removals and mitigation potential, and identify and prioritise the knowledge gaps and uncertainties that most limit the reliability of the LULUCF component of forecasts of New Zealand's net emissions position.

3. Objectives

3.1 Forest Sinks and Mitigation Options

The first Objective was to critically review, analyse, and summarise key data on New Zealand's forest biomass carbon stocks, rates of change, and estimate the associated level of uncertainty. We further aimed to develop current best-available forest biomass data with which to determine the effects on New Zealand's net position of likely post-2012 LULUCF activities, mitigation opportunities, and accounting options by:

- using existing datasets and models to estimate time series of exotic and indigenous forest-biomass stocks from 1985 to the present, including estimates of the area of land-use change by land class and previous vegetation cover;
- extending these data to at least 2020 under a range of likely future planting/regeneration, harvesting, and deforestation scenarios. This included analysis of the potential for increased use of forestry for emissions mitigation and erosion control on marginal lands;
- estimating the increase in biomass carbon stocks that could be achieved through improved management or other accountable activities post-2012 in exotic and indigenous forests. This included estimates based on IPCC or international data where New Zealand data were not available;
- providing, where possible, a statistical assessment of uncertainty and estimating the likely ranges of values when formal assessment was prevented by critical knowledge gaps.

3.2 Soil Carbon and Mitigation Options

The second Objective was to critically review, summarise, and document the effects of land use, land-use change, and mitigation options on New Zealand soil carbon stocks and rates of change and use the best currently available soil carbon data to determine the effects of likely post-2012 LULUCF activities on New Zealand's net position, mitigation opportunities⁴, and accounting options by:

- identifying and cataloguing the key characteristics, strengths, limitations and gaps in existing soil carbon datasets that are available to support the future development of soil carbon inventories for post-2012 LULUCF activities, mitigation opportunities and accounting options;
- generating from existing analyses of these datasets the best current estimates of soil carbon stocks and their change in forest land, grassland and cropland from 1985 to the present as a result of changes in land use and management (including intensification);

⁴ We do not propose to estimate in this proposal the possible magnitude of emissions mitigation opportunities available through use of biochar in agricultural, cropping or forest soils. However, we have commented, where appropriate, on the suitability of land-use and management practices to include a biochar component, the circumstances in which application of biochar may be particularly advantageous or disadvantageous, and the issues related to competition for the material used as possible biochar feedstocks, as potential bioenergy sources or to maintain viable soil function.

- extending these estimates to at least 2020, under a range of likely future land conversion rates, mitigation options based on changes in management practice, possible future;
- constraints on nutrient and irrigation application rates, and the likely effects of climate change;
- quantifying the net effect of erosion, and soil recovery on erosion scars, on national soil carbon stocks, including identification of likely anthropogenic and non-anthropogenic components and any mitigation opportunities;
- providing, where possible, a statistical assessment of uncertainty, and estimation of likely ranges of data where a formal statistical assessment is prevented by critical knowledge gaps.

3.3 Implications of LULUCF Accounting Options

The third Objective was to determine the implications of post-2012 accounting and mitigation options for net removals by the LULUCF sector, and to identify and prioritise the key knowledge gaps and uncertainties that most limit the reliability of the LULUCF component of forecasts of New Zealand's net emissions position by:

- using the best available time series of data on carbon stocks in forests and soils, from Objectives 1 and 2, to determine the contribution of the LULUCF sector and mitigations options to New Zealand's net position under post-2012 accounting options agreed with MAF;
- determining the effect that data uncertainty and knowledge gaps in the LULUCF sector have on the reliability of net position forecasts;
- proposing a prioritised work plan to resolve critical uncertainties and knowledge gaps for accounting in the LULUCF sector.

4. Methodology

4.1 Forest Sinks and Mitigation Options

For this Objective, we completed and documented a comprehensive review and synthesis of the information available to construct past, present and likely future changes in forest area, and forest-biomass carbon stocks and their rates of change. This has included consideration of emissions mitigation options based on improved forest management, and expanded use of forestry on marginal and erosion-prone lands. Most of the work has involved summarising and synthesising data drawn from existing New Zealand published and unpublished scientific investigations, models, reports, and datasets. It also includes an evaluation of the strengths, limitations, uncertainties and gaps in these existing datasets. Information from international work and reviews has been used to support our conclusions and generalisations where possible, especially when New Zealand data are limited.

In the present review⁵ we:

- conducted a critical review and synthesis of rates of change in exotic and indigenous forest biomass stocks from at least 1990 to the present, presented separately for both pre-1990 and post-1990 forests, and including indigenous shrublands;
- assessed the implications of future planting, harvesting, deforestation and management scenarios on future forest biomass stocks;
- assessed various mitigation options including the potential for emissions reductions from improved management of exotic and indigenous forest, changes in harvesting practice, and the expanded use of forestry on marginal and other lands.

We have examined the data from this review for reliability and assigned statistical measures of uncertainty where this was possible. Where uncertainty could not be formally quantified, we aimed to assign the probable range qualitatively in values of key parameters.

To improve the reliability of our conclusions, some additional work was undertaken to update existing data sets and remove known limitations or errors. This was largely related to improving estimates of rates of change of carbon stocks in indigenous forests (as might be required for net-net accounting of all forest lands), and of carbon stocks achievable through afforestation/reforestation of marginal lands (by updating both the estimates of available land areas and sequestration rates).

This section of the report (Chapter 3 and 4) concludes with a summary of the present status of forest biomass datasets, makes forecasts of biomass change, in relation to post-2012 accounting and mitigation options, and assesses their likely effect on New Zealand's future net position. Where possible, the report tries to separate anthropogenic from non-anthropogenic effects on change over time in forest biomass stocks. The report also provides conclusions on the effects of uncertainty and knowledge gaps on the reliability of analyses and forecasts, and provides a prioritised plan for reducing uncertainty and removing key knowledge gaps.

4.2 Soil Carbon and Mitigation Options

For this objective, we completed and documented a comprehensive and critical review of information available to estimate past, present, and likely future variations soil carbon stocks on forest land, cropland and grassland. Where possible, carbon stocks and changes for these land types has been further disaggregated according to land-management practice and related mitigation options. Changes in soil carbon as a result of land-use change between forests, pastures and crops, together with the effects of erosion, are treated as cross-cutting issues.

The major topics covered in the review are the impacts on, and mitigation opportunities for, soil carbon stocks and their changes in relation to:

- forest soils after afforestation/reforestation or deforestation, or in response to harvesting and variations in management;

⁵ A comprehensive review of the accounting of harvested wood products (HWP) is not included in the present work due to continuing uncertainty as to likely accounting options, and because it is recognised that this work may be being completed under other contracts. However, the key conclusions from current work for MAF under other contracts have been included here for completeness.

- cropping and horticultural soils under conventional tillage, low/no tillage and in response to other mitigation options, or to intensification of land use, including increased irrigation and fertiliser applications, liming, forage cropping, and use of organic farming practices;
- pastoral agricultural soils under extensive and intensive management (including enhanced irrigation and fertiliser inputs and liming), pasture renewal, and tussock grassland reversion;
- the effects of change in land use between forest land, cropland and grassland;
- the effects of erosion.

We have examined the data available from this review for their reliability and assigned statistical measures of uncertainty where possible. Where uncertainty could not be formally quantified, we aimed to assign the probable range qualitatively in values of key parameters.

To improve the reliability of our conclusions, some additional work was undertaken to update the existing data and remove known limitations or errors. This was largely related to improving existing estimates of national erosion rates and soil recovery, and to separating anthropogenic and non-anthropogenic components. This has allowed better estimates of the accountable net soil carbon balance for New Zealand under post-2012 accounting options. We have also completed a re-analysis of national soils data by removing possible existing sources of bias in order to provide improved stock-change factors for afforestation/reforestation.

This section of the report concludes with a summary of the present status of soil carbon datasets, and forecasts of soil carbon change, in relation to post-2012 accounting and mitigation options and the likely effect on New Zealand's future net position. Where possible, the report separates anthropogenic from non-anthropogenic effects on changes in soil carbon stocks. It also provides conclusions on the effects on uncertainty and knowledge gaps on the reliability of our analyses and forecasts, and provides a prioritised plan for reducing these uncertainties and removing key knowledge gaps.

5. Implications of LULUCF Accounting Options

For this objective, we have developed an integrated analysis of the effects of likely post-2012 accounting options for the LULUCF sector on New Zealand's net position under an expected and realistic range of future LULUCF activities and mitigation options. This work was based on the datasets, forecasts, mitigation options and measures of uncertainty developed in Objectives 1 and 2. The following accounting options were analysed:

- A status quo (Gross-Net) approach to LULUCF accounting.
- An All-Lands Gross-Net Accounting approach.
- An All-Lands Gross-Net Accounting approach with an additional cap on credits/debits from forest management.
- An all-lands net-net accounting approach referenced to 1990.

- A forward-looking baseline approach.
- The Average Carbon Stocks (ACS) approach.

The implications of overall uncertainties and knowledge gaps for forecasts of the contribution of the LULUCF sector to New Zealand's net position under the various accounting options are quantified to the extent possible. This information was used to produce a prioritised plan for addressing those areas that most limit the reliability of forecasts. The analysis has been segregated by LULUCF sub-sectors that are considered "key categories" under UNFCCC reporting, to provide information on the magnitude of uncertainty with respect to both carbon stocks, and rates of change in stocks—consistent with a key categories analysis. This will further help prioritise future work to address knowledge gaps and uncertainty.

CHAPTER THREE EXOTIC FOREST SINKS AND MITIGATION OPTIONS

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Summary

This chapter deals with the contribution New Zealand's planted forests make to carbon stocks, and stock changes, reported and accounted as part of the LULUCF sector under the United Nations Framework Convention on Climate Change (UNFCCC) and the Kyoto Protocol.

Planted exotic forests in New Zealand are estimated to provide about 84 Mt CO₂ of removal units during the first commitment period (CP1) and will thereby make a substantial contribution to New Zealand's ability to meet its Kyoto targets over CP1. Deforestation, primarily from pre-1990 exotic forests, is currently estimated to contribute a liability of about 17 Mt CO₂ over CP1. Whether a liability of that magnitude will actually eventuate may depend largely on the extent to which deforestation will be discouraged through policy means, such as the ETS, in future years. The contribution planted exotic forests can make as net carbon sinks over future commitment periods depends strongly on trends in future rates of both afforestation and deforestation as well as on future accounting rules.

The scope of the present study is to:

- use existing data and models to estimate and forecast planted exotic forest carbon stocks and change until at least 2020, under likely post-2102 accounting rules
- evaluate changes in carbon stocks under a range of likely future planting, harvesting, and deforestation, and forest management scenarios—including increased use of forestry for emissions mitigation and erosion control on marginal lands
- document the key strengths, limitations, critical assumptions, uncertainties and knowledge gaps involved in making estimates and forecasts of planted exotic forest carbon stocks
- identify and prioritise the knowledge gaps and uncertainties that most limit the reliability of the LULUCF component of forecasts of New Zealand's net emissions position.

The following are essential features of New Zealand's planted exotic forests in relation to carbon reporting and accounting:

- The total forest area is about 1.80 million hectares (Mha), with Radiata pine (*Pinus radiata*) comprising about 89% of the total area and with Douglas-fir (*Pseudotsuga menziesii*) the next most common species at just 6%. About one-third of these forests have been established since 31 December 1989, and are thus potential "Kyoto forests".
- Existing estimates and forecast of carbon stocks in planted exotic forests are presently based on data from the National Exotic Forest Description (NEFD). In future, it is likely estimates of planted forest carbon stocks will be made using a plot-based sampling approach in conjunction with LiDAR data, under the Ministry for the Environment's (MfE's) LUCAS inventory programme. However, the relationship between these two sources of estimates remains uncertain.
- The key strength of the NEFD-based approach is the wealth of age-class data it contains. This allows forecasts to depict accurately the changing effects of forest age class, which have a very large impact on total carbon stocks and change. The primary weakness of the approach is that NEFD yield tables (of volume by age) may underestimate wood volume in post-1990

forests, which tend to be on more fertile sites, and so grow more rapidly—although such effects are likely to be compensated to a substantial degree by wood on more fertile sites having a lower wood density.

- Forecasts of carbon stocks and change in planted exotic forest show that under a gross-net accounting regime, with a 1990 baseline year for determining eligible forest land, and without consideration of the fate of wood products:
 - at present planting and deforestation rates, post-1989 carbon stocks will continue to rise until about 2020 and then decline for about a decade, which means that these forests will become a source of emissions during that time
 - a minimum planting rate of about 65 000 ha per year is required to prevent post-1989 exotic forests from becoming a net emissions source in any five-year commitment period (CP). With a planting rate of 20 000 ha per year, post-1989 forest stocks will still decline from about 2020, and by about 2030 will reach a minimum of about 70% of 2020 levels. Carbon stocks will increase again after that.
- It is likely New Zealand could gain significant additional forest sink credits during CP1 from a downward revision of the estimated percentage of over-planting of indigenous shrubland (i.e. exotic forests planted on land that previously contained forests so that it would not be classed as land-use change and could not generate eligible removal units), and of soil carbon losses associated with afforestation of grassland—particularly in eroded landscapes. Progress in improving data quality in these areas has been very slow.
- Any move to all-forests net-net accounting with a 1990 baseline year, or a multi-year baseline centred on 1990, will be strongly disadvantageous to New Zealand. Pre-1990 forest removals peaked around that time, and all-forest removals peaked about 2006. Unless a very substantial expansion in future planting occurs, planted forests in New Zealand would therefore be accounted as a net emissions source beyond CP1 under most circumstances. By way of comparison, accounting during CP2 (taken as 2013–2018) is forecast to result in net accountable *removals* by forests of about 77 Mt CO₂ under the same accounting regime as applied during CP1, but net accountable *emissions* of about 10 Mt CO₂ under an all-forests net-net accounting regime with a 1990 baseline. Forecasts over a longer term, out to 2050, show continuing substantial disadvantage to New Zealand under an all-forests net-net accounting regime, even under scenarios involving average new planting rates of 20 000 ha per year.
- The following forest management options could be considered to increase forest carbon sinks under the current gross-net or any form of net-net accounting rules:
 - Increasing rotation age
 - Use of alternative species, especially those suited to longer rotations
 - Increasing volume and/or wood density through genotype selection and/or modifying stocking.
- Retaining the pre/post 1990 split for planted exotic forests, and accounting carbon stocks in pre-1990 forests on a net-net or forward-looking-baseline basis—with post-1990 forests accounted on a gross-net basis—may be an option New Zealand wishes to further consider. Such a split-accounting approach appears to be substantially more favourable than an all-forests net-net accounting or forward-looking baseline approach—though still considerably less favourable than the CP1 gross-net (post-1990) plus deforestation-liability (pre-1990), approach. More work would need to be done on a split-accounting option to determine precisely the implications for New Zealand if the option were to become a serious international contender, as it was not considered in detail in this study.

- For a split-accounting approach to have the most favourable outcome for New Zealand, incentives would need to be introduced to:
 - minimise liabilities from deforestation of pre-1990 forests (including through advancing the concept internationally of “land-use flexibility mechanisms⁶”
 - minimise liabilities from deforestation of post-1990 forests (including through extension of the “fast forest fix⁷” to cover both harvesting and deforestation of post-1990 forests)
 - adopt forest management options for forests subject to net-net accounting as outlined above—extending rotation time in particular. International adoption of New Zealand’s proposed “land-use flexibility” mechanism for pre-1990 forests would greatly assist in ensuring pre-1990 forests accrued, at the very least, no liabilities
 - plant at least 20 000 ha of new forest annually onto grassland to avoid post-1990 forests becoming a net source at some point in the future.
- Various analyses have shown that there are at least 1 Mha of marginal, severely erosion-prone, pastoral lands that would benefit from exotic afforestation—and potentially as much as 2.6Mha if moderately erosion-prone lands are included as well. This provides sufficient scope to support a planting programme, with associated environmental co-benefits, of at least 20 000 ha per year until 2050. Carbon accumulation on such lands is expected to average about 25 t CO₂ per ha per year. The marginal lands are in the following ownership classes:
 - Crown-owned land: 0.13 Mha (or 0.25 Mha⁸)
 - Privately owned land: 0.92 Mha (or 2.36 Mha)
 - Maori-owned land included in privately owned land: 0.05 Mha (or 0.11 Mha).

The major knowledge gaps and uncertainties that most limit the reliability of estimates and forecasts of the contribution planted exotic forest net removals make to New Zealand’s net emissions position are (in order of priority):

- The most likely post-2012 accounting regime—this has by far the largest impact on the modelling and forecasting of forest emissions/removals, with *all-forests* or net-net approaches being very disadvantageous to New Zealand compared with CP1 accounting rules.
- Limitations of the NEFD-based approach to modelling in terms of applicability to post-1989 forests (effect of soil fertility and unknown management regimes), and inability to provide estimates with statistically-based confidence limits. Inventories completed under MfE’s LUCAS approach may address some of these issues. There are also calibration/validation issues to be addressed with the proposed LUCAS approach itself, but these are beyond the scope of this report.
- An urgent requirement is for available LUCAS data for planted forests to be analysed and used to calibrate or replace both the post-1989 and pre-1990 NEFD-based yield tables. Verification of NEFD and other planted forest areas from LUCAS mapping work, if sufficiently reliable, is also a priority.

⁶ The land-use flexibility option being advanced by NZ allows for an area of pre-1990 exotic plantation forest to be harvested, and replanted in another location, without being considered deforestation.

⁷ The “fast-forest fix” states that debits resulting from harvesting during the first commitment period following afforestation and reforestation since 1990 shall not be greater than credits accounted for on that unit of land.

⁸ The alternative, larger areas are those that result if lands with a moderate erosion risk are included.

- Improved characterisation of the rates and forecasts of deforestation and harvesting. This includes effects on the soil carbon pool, especially if areas of land-use change become larger under (currently proposed) land-use flexibility mechanisms. It also includes development of least-cost remote sensing techniques (probably based on satellite radar imagery) to map deforestation and harvesting.
- Development of carbon partitioning and wood density functions for Douglas fir, and to a lesser degree for eucalypts.
- Development of reliable defaults for understorey carbon stock changes following planted exotic afforestation, and for emissions due to pre-afforestation land clearance, as a function of climate/soil conditions. Net-net accounting may also require estimation of pre-afforestation sequestration rates, depending on the form of the baseline approach.
- Validation of assumptions in relation to the proportion of carbon stocks removed off-site and residues remaining, particularly if residues become a source material for biochar or bioenergy.
- Development and validation of indices related to forest management activities in planted forests that can be determined using remote sensing techniques.

1. Introduction

Increasing carbon stocks in exotic plantations established after 1990, and carbon losses from some of these plantations that are now being converted back to pasture, constitute the quantitatively most important accountable carbon-stock changes from New Zealand's biosphere. This Chapter is therefore dedicated entirely, and specifically, to the latest quantification of carbon stock changes with land-use change from pasture to exotic forest and vice versa. It also presents extensive simulations of the effect of various accounting options on New Zealand's net position over future Commitment Periods under the Kyoto Protocol.

1.1 Scope of Study

In this chapter we provide estimates and forecasts of carbon stocks and stock changes on planted forest land, from 1990 to at least 2020. Separation of forests into planted forests and natural forests is not a reporting requirement under the UNFCCC, although it is good practice to stratify forests if this will improve the accuracy and transparency of inventory estimates—which is the case for New Zealand. Planted forests are also separated because post-1990 forest sinks, which are almost exclusively planted exotic forests in New Zealand, make a key contribution to New Zealand's emissions offsets during CP1. (Indigenous forests are considered in Section 2 of this chapter.)

The study reviews the status of existing data and models used to estimate and forecast carbon stocks in the four IPCC-defined biomass-related pools of New Zealand's planted exotic forests: above-ground biomass, below-ground biomass, dead-wood and litter. The study also documents the key strengths, limitations, critical assumptions, uncertainties and knowledge gaps involved in making estimates and forecast of planted forest carbon stocks. The existing forest carbon stock models have been used to develop scenarios to illustrate the impact of changes in assumptions on carbon pool projections. Soil carbon in planted exotic forests is addressed more completely in Chapter 5, and is covered in this chapter only in the context of describing the scope of existing national inventory estimates and projections.

As at April 2006, the total area of planted exotic forests in New Zealand was 1.80 million hectares (MAF, 2007). Radiata pine (*Pinus radiata*) was by far the most common plantation species, with about 89% of the total area—and is thus the almost exclusive focus of this study. Douglas-fir (*Pseudotsuga menziesii*) is the next most common species at just 6%. About one-third of these forests have been established since 1 January 1990.

Non-CO₂ greenhouse gas emissions due to burning of forest biomass burning, although included in the national greenhouse gas inventory prepared under the UNFCCC, are very small compared with forest carbon stock changes and are not reviewed here.

Existing estimates of carbon stocks in planted forests have effectively used the National Exotic Forest Description (NEFD) survey definition of planted forests. The NEFD survey compiles areas by age class for planted production forests, defined as “an area of trees, not less than one hectare in size, planted and managed with the intention of producing wood or wood fibre” (NEFD 2007). This potentially includes planted stands of indigenous tree species and naturally seeded stands of exotic species, although neither are at all common. However, it excludes shelterbelts, amenity plantings, regenerating natural forests and wildling pines.

In future it is likely estimates of planted forest carbon stocks will be made using a sampling approach, where planted forests are defined according to the requirements under the Marrakech Accords. For the thresholds of forest area, height, and crown cover adopted by New Zealand, the forest definition under the Accords is:

“Forest” is a minimum area of land of 1.0 hectare with tree crown cover (or equivalent stocking level) of more than 30 percent with trees with the potential to reach a minimum height of 5 metres at maturity in situ. A forest may consist either of closed forest formations where trees of various storeys and undergrowth cover a high proportion of the ground or open forest. Young natural stands and all plantations which have yet to reach a crown density of 30 percent or tree height of 5 metres are included under forest, as are areas normally forming part of the forest area which are temporarily unstocked as a result of human intervention such as harvesting or natural causes but which are expected to revert to forest.

As well, the forest must be greater than the New Zealand adopted width of 30 metres. This additional restriction excludes windbreaks and some riparian forest areas, which are common on New Zealand agricultural lands. Overall, this forest definition will result in coverage being extended to include stands where, unlike the NEFD definition, the management objective is not necessarily wood production—for example, poplars planted for erosion control or amenity plantings. Wildling conifers are also likely to be included if they exceed the thresholds adopted by New Zealand for the definition of forest under the Marrakech Accords. Post-1990 forest that meets the definition given under the Marakech Accords is frequently referred to as Kyoto forest, because of its eligibility during CPI as a carbon sink under the Kyoto Protocol.

1.2 Reporting and Accounting Requirements

Article 3.3 of the Kyoto Protocol allows the net changes in greenhouse gas emissions by sources and removals by sinks resulting from afforestation, reforestation and deforestation (ARD) since 1 January 1990 to be used to meet New Zealand’s commitments. Emissions and removals must be measured as verifiable changes in carbon stocks in each commitment period. The carbon stocks to be accounted are above-ground biomass, below-ground biomass, dead wood, litter, and soil organic carbon. Emissions and removals from these same pools must also be reported under the UNFCCC in New Zealand’s annual greenhouse gas inventory. However, the UNFCCC inventory reports carbon stocks in all forests (i.e. including pre-1990 forests) as part of the Land Use, Land Use Change and Forestry (LULUCF) sector.

The Land Use and Carbon Analysis System (LUCAS) is being developed and implemented by the Ministry for the Environment, so that New Zealand can meet its international obligations for reporting under the Kyoto Protocol. The basis for estimating stock changes in forest biomass and dead organic matter pools will be a representative sample of plot measurements taken from the Kyoto Forest estate at or near 1 January 2008 and repeated at 31 December 2012⁹. A combination of field measurements and airborne LiDAR (Light Detecting and Ranging) will be used across a 4-km grid. The field measurement programme commenced in 2007 and it is intended to repeat this at the end of the commitment period, and to extend the programme to cover pre-1990 planted forests.

Because a national coverage of LUCAS carbon monitoring plots has yet to be established, an alternative approach using NEFD data has been used to date to estimate net CO₂ removals for the contribution of planted forests to greenhouse gas inventory and projections during CPI. A key benefit of the NEFD data is that it provides planted forest areas by age class. This age class

⁹ Either measurement may be replaced with modelled stocks if necessary.

information is crucial to projecting future (and past) stocks, and is not readily available from remotely sensed data.

Inventory reporting will rely on the data and models described in this report until LUCAS is able to provide estimates from nationally representative plots.

2. Estimating Planted Forest Carbon Stocks

2.1 Approaches to Accounting

Various approaches have been used in New Zealand to estimate historic and future forest carbon stocks and the change expected during Kyoto commitment periods. The IPCC inventory guidelines broadly classify approaches according to a hierarchical tier structure, with higher tiers implying increased accuracy of the method and/or emissions factors and other parameters used in the estimation of emissions and removals.

The three tiers are:¹⁰

- Tier 1: uses the basic methods provided in the IPCC Guidelines, with default emissions factors and usually spatially coarse activity data
- Tier 2: may use the same method, but applies country-specific emission factors and activity data for the most important land uses/activities
- Tier 3: applies higher order methods including models and inventory measurement systems tailored to address national circumstances, repeated over time, and driven by high-resolution activity data. Such systems may be GIS-based combinations of age, class/production data systems with connections to soil modules, integrating several types of monitoring. Pieces of land where a land-use change occurs can be tracked over time.

2.1.1 Tier 1 analysis

This is the simplest approach, based on average stocks and changes in total area by land use (*GPG-LULUCF*). Such a Tier 1 analysis is carried out each year as part of the UNFCCC Greenhouse gas inventory. A land-use change matrix has been developed from a comparison of two land cover databases—LCDB1 (compiled in 1997) and LCDB2 (compiled in 2002). Changes before and after this time are estimated by linear extrapolation of the trend between 1997 and 2002, meaning that, among other things, the more recent sharp decline in afforestation and increased conversions of plantations to agricultural land are not reflected.

The use of average carbon stocks also causes problems, as carbon stocks in planted forests are strongly correlated with age. The age class distribution is therefore of importance—for example, in the 1970s the New Zealand planted forest resource was expanding in area but was still a net carbon source due to harvesting and replanting of the over-mature ‘old crop’ (MacLaren et al. 1995).

Tier 1 analyses are not appropriate for key categories in a national greenhouse gas inventory, which includes forest land in New Zealand. For this reason, the Tier 1 analysis results are only retained for the non-forest land uses. The forest land estimates are replaced with the results of a Tier 2

¹⁰ Box 3.1.1 Framework of Tier Structure in Good Practice Guidance (*GPG-LULUCF*)

analysis, based on more detailed modelling using area and yield data from the National Exotic Forest Description (NEFD).

2.1.2 Tier 2 analysis

New Zealand analyses using a Tier 2 approach include the forest modelling for the UNFCCC greenhouse gas inventory (e.g., Wakelin 2007, MFE 2008a), projections of the “Kyoto net position” (e.g., Wakelin et al. 2008b, MFE 2008b), and earlier estimates.

The earliest estimates of carbon stocks in New Zealand planted forests were probably those of MacLaren and Wakelin (1991). These were based on NEFD area and yield data, with annual activity data (e.g., planting and harvesting) simulated using the FOLPI estate modelling system (Garcia 1984). This is essentially the same approach used for New Zealand’s annual greenhouse gas inventory reporting since that time.

Another early attempt at quantifying carbon sequestration was made by Hollinger et al. (1993), also using NEFD area data. In this exercise, the base year data was only projected for a single year, so a spreadsheet was used rather than specialist estate modelling software. An Excel spreadsheet model has also been used for estimating New Zealand’s Kyoto net position for the LULUCF sector (Wakelin et al. 2008b). In this case, the need for more complex manipulation of age classes was avoided by assuming that no harvesting of post-1989 forest would occur before or during CP1.

In all the Tier 2 analyses mentioned, NEFD stand volume yield tables were converted to stand carbon yield tables using the relationships contained within the Drymat (Beets 1982) and C_Change (Beets et al. 1999) models. When combined with the current age class distribution, the current carbon stock can be calculated. The effects of future activity (e.g., afforestation, reforestation and deforestation) are then simulated over time and future stocks calculated by matching expected areas with yields at the appropriate ages.

2.1.3 Tier 3 analysis

It is unlikely national planted forest carbon estimates have been produced for New Zealand using a Tier 3 approach, although it is possible individual forest owners have carried out such analyses on their own resources.

The Tier 2 forecasts and their underlying data and models are reviewed in the following sections, as they provide the most up-to-date projections of carbon stocks in both post-1989 and pre-1990 forests. However, it should be recognised that while the NEFD-based approach has been used for UNFCCC reporting in the past, LUCAS is being designed to provide robust LULUCF sector inventory data specifically for Kyoto Carbon Accounting purposes. The approach and much of the data described in the following sections are therefore not expected to be used to calculate and report on New Zealand’s CO₂ removals from planted forests in the future.

2.2 Overview of Latest Forecasts

2.2.1 Kyoto net position model

New Zealand’s projected quantity of emissions and removals of greenhouse gases during the first commitment period of the Kyoto Protocol is updated annually by MFE (MFE 2008b). Projections for the agricultural and LULUCF sectors are provided by the Ministry of Agriculture and Forestry. The LULUCF sector analysis is limited to planted forest ARD activities since 1990—it is currently

assumed indigenous forests are neither a source nor a sink. Spreadsheet modelling of emissions and removals from planted forests is carried out by Scion under contract to MAF (Wakelin et al. 2008b).

Outputs from the modelling include annual emissions and removals associated with ARD activity from 1990 to 2012. The model can provide projections to 2052 if ARD scenarios are available beyond 2012. The components that can be quantified include:

- afforestation removals, with reduction of post-1989 forest area due to the fact that some areas may have been planted onto land that already met the definition of forest
- afforestation emissions, as a result of clearing existing woody (but non-forest) vegetation
- afforestation emissions associated with soil carbon losses
- deforestation emissions.

However, removals and emissions from forest stands that already existed in 1990 are not modelled, except in the case of deforestation of post-1989 forests. This means the “Kyoto Net Position” model cannot be used as a basis for projecting emissions and removals under more comprehensive post-2012 accounting rules.

2.2.2 UNFCCC GHG inventory model

The latest National inventory report (MFE 2008a) and common reporting format tables are available from the IPCC website.¹¹ An annual GHG inventory is provided from 1990 to 2006, with removals and emissions in planted forests provided by MAF from FOLPI modelling carried out by Scion (Wakelin 2007).

While only the 1990–2006 results are reported, the modelling extends from 1980 to 2060. It is also possible to report separately on pre-1990 and post-1989 forests. This means there is potential to conduct analyses of alternative post-2012 accounting rules using the same model used as the basis for the currently reported inventory.

The projections for the post-1989 forests made using the UNFCCC inventory model are generally consistent with the Kyoto Net Position model, as the same base MAF planting data and carbon yield table is used. However, all post-1989 planted forests are included, with no reduction in area due to potential over-planting. On the other hand, there is more scope to explore scenarios that vary factors such as changes in rotation length in both pre-1990 and post-1989 forests.

2.2.3 Other models

The future growth of LUCAS PF CMS plots can be forecast using the 300Index growth model and C_Change. Some initial analysis of this data has been carried out, but no attempt has been made to provide national estimates due to the limited data coverage. The UNFCCC inventory model described above has been used as the basis for modelling the whole planted forest resource under scenarios that vary in rotation length (Wakelin 2008). These models provide the basis for future projections of pre-1990 planted forest CO₂ removals reported here.

¹¹ http://unfccc.int/national_reports/annex_i_ghg_inventories/national_inventories_submissions/items/4303.php

2.3 Additional Requirements for Post-2012 Accounting Options

The models described above are tailored towards the current reporting requirements. Any change in the post-2012 Kyoto accounting rules are likely to require changes in the data and model. Current estimates do not “factor out” the influence of CO₂ fertilisation, nitrogen deposition and global warming effects, but these effects are not included in the models in any case—productivity is assumed to be independent of such factors. There may be a requirement to separate direct human-induced influences from natural events such as pest outbreaks, windthrow and fire, or to separate future removals associated with a “business as usual” scenario from additional removals due to “improved management”. Harvested Wood Product accounting may be required under post-2012 rules. Net-net accounting places increased emphasis on baseline net removals, and a methodology for factoring out “age class effects” is required. Discussion of data and model limitations in the following sections are based on assumptions about likely accounting approaches.

3. Review of Data and Models

3.1 Approach to Date

The modelling approach taken to date has been based on the use of:

- NEFD areas and MAF estimates of activity data (ARD and harvesting)
- NEFD stem volume yield tables
- C_Change relationships to convert stem volumes to stand carbon by pool.

By international forestry standards, New Zealand has a high level of knowledge regarding its intensively managed planted forests, and relatively good data. However, many of the data sources used were not designed for carbon accounting purposes. The key data elements are discussed in turn, followed by an assessment of the models themselves.

3.2 Activity Data

Planted forest areas are taken from the National Exotic Forest Description (NEFD) database (e.g., MAF 2007). This combines data from a survey of major forest growers undertaken by the Ministry of Agriculture and Forestry, a Small Forest Growers survey completed by AgriQuality in 2004, and estimates of new planting based on data obtained from nursery surveys (Eyre, 1995). MAF has estimated the total area of planted forests in the NEFD to be accurate to $\pm 5\%$.

Some of the issues surrounding the use of NEFD area data are discussed in Wakelin (2005) and Wakelin (2008). While the total area may be reasonably accurate, activity data are required at a higher level of resolution, including:

- historic and projected future afforestation, including identification of land planted with existing vegetation (both Kyoto-compliant and non-compliant)
- historic and projected future harvest areas
- historic and projected future deforestation.

3.2.1 Historic afforestation

Ideally afforestation estimates would be available to cover both pre-1990 and post-1989 forests, and broken down by NEFD crop-type (region/species/regime combination) or alternative attributes that can be used to determine an appropriate yield table.

The NEFD survey is now able to characterise 94% of the resource into first or later rotations. This is important for assigning appropriate post-harvest residues and improving the characterisation of the post-1989 resource, and has been used in this report to test the appropriateness of the national carbon yield table used in previous stock estimates.

Afforestation estimates from NEFD survey returns are topped up with estimates inferred from the nursery survey. This requires estimates of the amount of replanting, blanking (re-planting where establishment has failed) and assumptions on the stocking at establishment. A review determined that the latter assumption was critical (Manley et al. 2003). Seedlings are not necessarily planted out in the regions in which the nursery is located, so the existing planted forest area is used to allocate afforestation to regions on a pro-rata basis. About half the post-1989 afforestation has been imputed in this way.

It is believed small owners may report gross area rather than net stocked area in the NEFD, and are less likely to adjust areas as a consequence of remapping stand gaps. In wood-availability models, these areas have been decreased by 15% (e.g. MAF 2006).

Afforestation also needs to be classified according to previous land use. This is for two purposes:

- To separate Kyoto-compliant afforestation from “forest over-planting” – that is, the afforestation of land that already met the definition of forest.
- To allow any carbon emissions associated with removal of existing vegetation to be estimated.

The available data for these purposes are poor. The “over-planting” issue has been modelled in the Kyoto balance of units calculations by assuming that a fixed proportion of annual afforestation (both historic and future) is ineligible under Kyoto rules due to the presence of forest before conversion. “Worst-case”, “Most Likely” and “Best-Case” assumptions of 21%, 16% and 8% were used. The Most likely and Worst-case values were based on the use of two national classifications to test the representativeness of a pilot mapping project in Nelson-Marlborough, in terms of post-1989 exotic forest planted into possible forest land. The two sources of data were the 1987 Vegetation Cover Map and the 2001/02 Land Cover Database. Spatial intersection of these indicated the likely area of post-1989 forest planted into possible forest land being: nationally 16%; Marlborough region 21%; and the Gisborne region 15%. Some anecdotal information at the time suggested that the levels could be as low as 8-10%, and this was used for the optimistic figure. (Peter Stephens, MfE, pers. comm.).

Net removals are less sensitive to assumptions about the clearance of previous vegetation prior to afforestation. The NEFD survey does capture previous land use for afforestation returns, and the proportion planted onto scrub has been used to estimate emissions from scrub clearance. However, the survey returns for this question are incomplete, and calculating appropriate emissions is difficult (Wakelin 2005). Afforestation of pasture is assumed not to result in land clearance emissions, and no conversion of native forest has been reported since 1990.¹²

¹² 28 000 ha of native forest were reportedly cleared for plantation from 1980 to 1988 (Forestry Facts and Figures 1990 NZFOA Inc).

Table 1 (below) gives a breakdown of afforestation since 1984, including the amount assumed to be planted onto cleared scrub and the amount assumed to be over-planted forest and therefore ineligible under Kyoto.

Table 1 Afforestation 1985–2007 (hectares)

Calendar Year	State	Private	Total	<i>Onto Scrub</i> *	<i>Onto Forest</i> **
1984	20 000	36 000	56 000	11 200	8960
1985	18 000	30 000	48 000	9600	7680
1986	15 000	25 000	40 000	8000	6400
1987	10 000	20 000	30 000	6000	4800
1988	3 000	17 000	20 000	4000	3200
1989	2000	19000	21 000	4200	3360
1990	0	15 800	15 800	3160	2528
1991	0	15 400	15 400	3080	2464
1992	0	50 200	50 200	10 040	8032
1993	0	61 600	61 600	9856	9856
1994	0	98 200	98 200	15 712	15 712
1995	0	73 900	73 900	8129	11 824
1996	0	83 600	83 600	11 704	13 376
1997	0	63 700	63 700	7007	10 192
1998	0	51 200	51 200	4608	8192
1999	0	40 000	40 000	5600	6400
2000	0	33 600	33 600	5712	5376
2001	0	30 100	30 100	6923	4816
2002	0	22 100	22 100	4199	3536
2003	0	19 900	19 900	5174	3184
2004	0	10 600	10 600	3922	1696
2005	0	6000	6000	4740	960
2006	0	2600	2600	1612	416
2007	0	1600	1600	714	256
Total new planting 1990–2007	0	680 100	680 100		

* From annual proportions reported in NEFD where available.

** From 16% 'Most Likely' assumption (Wakelin et al. 2008b).

3.2.2 Future projected afforestation

Ideally, projections of afforestation are required at a higher resolution than a simple national total, so that appropriate assumptions on growth and carbon removals can be made. The assumption in current models is that there is no difference in growth rate between pre-1990, existing post-1989, and future planted forests. Afforestation rates of 0, 5000 and 20 000–30 000 ha per year were modelled for the Kyoto balance of units, based on MAF scenarios (Wakelin et al. 2008b). This is sufficient for CPI analysis because new afforestation has little impact, but ideally more rigorous analysis would be undertaken when looking at longer term uptake trends. This would include specific analyses for likely sites available for afforestation.

3.2.3 Historic harvesting

The UNFCCC inventory model uses estimates of annual harvested areas to shift the latest NEFD age class distribution back to a 1980 start date. The estimate is derived from an estimate of clear-fell round-wood removals, the national average yield table, and an assumed rotation age. The clear-fell round-wood removals estimate is ultimately derived from mill production data, assumed mill conversion rates and an estimate of the proportion of round-wood supplied from thinning operations. While this process ensures the model will achieve the current age class distribution having simulated national afforestation, harvesting and restocking since 1980, the many assumptions mean that historic age class distributions and stock estimates are indicative at best.

In addition, modelling at a sub-national level would require national harvesting estimates to be apportioned to regions and ultimately crop-types.

3.2.4 Future projected harvesting

Key assumptions for determining annual future harvest areas include:

- rotation age
- timeframe over which currently mature stands will be harvested
- yield regulation constraints, e.g., maximum annual percentage increase in harvest level; non-declining yield.

Both the UNFCCC inventory model and Kyoto Net Position model assume a base rotation age of 28. In the Kyoto model this is fixed, so harvesting of post-1989 forests does not begin until CP3. In the UNFCCC model, the age is a target, and some fluctuation is allowed to meet yield regulation constraints. These constraints are applied as a surrogate for logistical considerations—eld is not permitted to decline and can only increase by up to 10% over the previous years harvest. Without these constraints, the very uneven age class distributions of the pre-1990 and post-1989 estates will result in extreme fluctuations of harvesting, and this has a direct impact on net removals.

3.2.5 Historic deforestation

Significant deforestation of planted forests is a recent phenomenon captured in NEFD surveys (MAF 2007). The Kyoto Net Position model currently assumes that deforestation takes place at age 28 in pre-1990 forests. To estimate the resulting emissions more accurately would require information on factors such as pre-deforestation carbon by pools, land conversion method, age and post-deforestation land use. Estimated and projected deforestation area estimates are given in Table 2.

3.2.6 Future projected deforestation

The Kyoto Net Position model uses projections of future national deforestation provided by MAF (Table 2; for 1990–2012). These are modelled in the same way as the historic deforestation areas—that is, at a national level, assuming deforestation follows normal harvesting at age 28. Projections of deforestation areas should ideally be characterised by factors relevant to emissions, e.g., region, age, site productivity, and species.

Table 2 Estimated and projected deforestation estimates

Calendar year	Base scenario ETS Policy (ha)	Optimistic scenario = ETS Policy (ha)	Pessimistic scenario No Policy (ha)	Smith and Horgan (2006)
1990-2003	1000	1000	1000	
2004	7000	7000	7000	
2005	12 900	12 900	12 900	
2006	12 700	12 700	12 700	12 700
2007	19 000	19 000	19 000	19 000
2008	2400	2400	7400	13 548
2009	2400	2400	7400	12 987
2010	2400	2400	7400	13 548
2011	2400	2400	7400	11 306
2012	2400	2400	7400	9811
2013–				9355 ha/year

3.3 Stem Volume and Yield Tables

The latest published set of NEFD yield tables date from 1995 (Ministry of Forestry 1996). Eighty-eight NEFD crop-type yield tables are available for combinations of region, species, regime and in some cases, radiata pine age cohort (pre- or post-1975). The yield tables are derived from yield tables supplied by the main forest growers in each region. A revised set is currently being developed in conjunction with MAF's wood availability forecasting exercise.

The NEFD yield tables were developed specifically for wood availability forecasting. LUCAS will replace the use of yield tables with representative plot data. However, until these data are available, both the UNFCCC greenhouse gas inventory and the Kyoto Net Position continue to use the NEFD yield tables. The advantages and disadvantages of using the NEFD yield tables in this way were discussed by Wakelin et al. (2005).

Using the NEFD yield tables was a pragmatic course of action: they already existed, with national coverage at a suitable level of resolution; formal and established mechanisms were in place through the NEFD steering group for ongoing support and maintenance; and the burden of validating, maintaining and improving the yield tables is shared by others. The other main advantage is that the yield tables have credibility, as they have had industry input and scrutiny and have been published. They are underpinned (at least in theory) by published growth models of long standing tempered by reconciliation with actual recoveries and inventory data.

The main problem with using the NEFD yield table as the basis for carbon yield tables arises from the relatively narrow focus of the NEFD Steering Committee. These yield tables were:

- prepared as a basis for wood availability studies, i.e. yields at the range of rotation ages modelled (20–40 but more typically 26–30 years)
- based mainly on areas that were expected to be harvested in the short- to medium-term from the large forest owners' resource in the late-1990s.

This means that:

- yields and increments may not be accurate at other ages—for instance, NEFD yield tables may over-predict yield (and therefore carbon) at young ages and radiata pine yield tables do not increment above age 40.
- yield tables do not necessarily reflect historic or future growth rates, and do not capture trends in productivity over time.
- yields may not be applicable to the small growers' estate.
- regime differences may not be explicitly captured in the yield tables at the time of silviculture, unlike yield tables produced using a stand growth simulation model.

Nevertheless, the NEFD yield tables are likely to remain the best published source of growth data suitable for national carbon modelling purposes until LUCAS is fully implemented, or wall-to-wall mapping of planted forests becomes available.

3.4 Carbon Yield Tables

The C_Change model (Beets et al. 1999) is used to convert stem volume to total biomass. C_Change is a compartment model for dynamically simulating the drymatter content of managed radiata pine stands and has been constructed from a large dataset of measurements from Puruki and elsewhere (Beets et al. 1999). Biomass data collection for construction of models or allometric equations is expensive and time-consuming. New Zealand is fortunate that radiata pine makes up 90% of the planted forest resource, allowing a concentration of effort.

The process used to derive a carbon yield table for each NEFD yield table is described in more detail in Wakelin et al. (2005). The two main steps are to:

- convert NEFD yields net of mortality to gross yields¹³
- use C_Change first to convert stem volumes to stem biomass, and then to convert stem biomass to stand biomass.

Inputs to C_Change include the NEFD stem volume yield tables, wood density classes for regions and species, and silvicultural regime details. C_Change is used to:

- derive stem wood biomass increment from volume increment and density
- apply an increment expansion factor to convert this to total carbon fixed
- partition the total carbon to live biomass pools
- calculate transfers from live to dead pools from mortality functions and regime details (i.e. pruning/thinning)

¹³ Gross yield is stem volume before thinnings and mortality are removed. C_Change requires both gross and net increments as inputs. Gross volume increment is used to calculate total dry matter production; the difference between gross and net volume is used to derive carbon in annual tree mortality, and the resulting dead carbon is added to the dead component carbon stock.

- apply decay functions to estimate carbon loss from dead pools.

The output from C_Change is a carbon yield table corresponding to each of the 88 NEFD crop-types, with estimates of carbon per hectare by age class for each pool. Note that these carbon yield tables assume:

- species-specific volume growth based on the species and species groups used in the NEFD yield tables
- broad wood density classes differentiated by species (and by region for radiata pine)
- regime assumptions (particularly initial and final stocking) based on radiata pine PSP data within each of the four recognised NEFD regimes
- carbon partitioning based on radiata pine relationships, as data for other species is limited.
- second rotations follow a 28-year rotation of radiata pine
- the carbon fraction (carbon as a proportion of oven-dry biomass) for each stand component is 0.5¹⁴
- the aggregations into carbon pools shown in Table 3.

Table 3 C_Change aggregations into UNFCCC inventory pools

Pool	Variable	C_Change variable description
Above-ground Living	X[3]	0–1 yr needle
	X[4]	1–2 yr needle
	X[5]	2+ yr needle
	X[6]	Live branch
	X[7]	Dead branch
	X[8]	Stem wood
	X[16]	Stem bark
Below-ground Living	X[9]	Coarse root
	X[14]	Live fine root
Dead-wood (coarse woody debris)	X[12]	stem litter
	X[13]	coarse root litter
Litter (fine woody debris)	X[10]	needle litter
	X[11]	branch litter

¹⁴ Note that while the IPCC default for the litter component of the Litter Pool is 0.37, initial investigations in New Zealand indicate that while this may be appropriate for the fomic and humic layers, it is too low for the litter pool as a whole once litter and fine woody debris are added (Haydon Jones, Scion, pers. comm.)

Note that in Table 3:

- [X15] fine root litter is considered to be included in soil carbon estimate
- the harvested pool taken offsite includes part of X[8] and X[16], calculated using the ratio of NEFD merchantable volume to total stem volume
- coarse woody debris includes stem litter and coarse root litter.

Validation of C_Change is provided by Beets et al. (2007).

3.5 Treatment of Soil Carbon

Soil carbon stock changes reported in the UNFCCC greenhouse gas inventory are obtained from the Tier 1 analysis. Both the Tier 2 inventory model and the Net Position model do include simple models of soil carbon change with changing land use. Soil carbon is assumed to:

- decline with the conversion of pasture to forest
- remain at forest levels following deforestation
- be stable if land use is unchanging.

Following deforestation, residues are assumed to be released back to the atmosphere on decay, with no transfers to the soil carbon pool. Possible changes to the soil carbon pool following deforestation and conversion to pasture are considered further in Chapter 5.

3.6 Models

3.6.1 The UNFCCC inventory model

The *UNFCCC inventory model* is based on the latest NEFD national age class distribution, which is projected both into the future and back to 1980 by simulating afforestation, harvesting, replanting and deforestation. This forward simulation process is robust, but is not required for inventory reporting, as only annual removals from 1990 to the current year are reported.

Five scenarios are modelled:

- Pre-1990 forest, rotation 1
- Pre-1990 forest, rotation 2+
- Post-1989 forest, rotation 1
- Post-1989 forest, rotation 2+
- Post-1989 restocking of pre-1990 stands.

Each scenario shares the same yield table—the separation is solely to allow stocks to be reported separately for pre-1990 and post-1989 forests, and to reflect post-harvest residues after the first rotation. The yield table is prepared by area-weighting the 88 carbon yield tables developed for the 88 NEFD crop-types described in Section 1.3.4 above.

Some simplifications are currently made:

- The harvested stem carbon is assumed to be instantly emitted, in accordance with the IPCC default approach.
- Harvest residues at the start of the second rotation are based on carbon pools at age 28 for an area-weighted national average radiata pine carbon yield table. The proportion of stem carbon removed at harvest is determined by the merchantable volume as a proportion of total volume in the yield table (about 85%). Because the formulation of FOLPI merges harvested stands at the time of replanting, initial residue levels do not necessarily reflect previous rotation characteristics. More accurate modelling of harvest residues would require each age class to be represented as a separate crop-type, with a separate replanting crop-type for each. At a national level, this would require 80 crop-types for each of the current five (i.e. 400 crop-types in total). If modelling was carried out at an NEFD crop-type level, the number of crop-types required would become prohibitive.
- Future deforestation is currently modelled as an instant emission within the inventory model (but future stocks and stock changes are not reported in the national inventory report)
- Afforestation is assumed to be onto land with no carbon present in biomass, except where planting onto scrub has been indicated. On these sites, an estimate is made of the emissions associated with scrub clearance.

The backwards simulation is not as straightforward, and requires a number of simplifying assumptions. Deforestation is not explicitly modelled—the latest NEFD age class distribution is already net of any stands harvested and deforested in the past. Harvested stands are ‘added back’ into the age class distribution in previous years by deriving the area planted from the volume harvested and the yield at an assumed rotation age. Deforestation areas that do not contribute to round-wood removal statistics (e.g. at young ages) are not added back – these areas are effectively assumed to have never existed. Estimates made for the 1990 base line and the time series to the present day are therefore more uncertain.

3.6.2 The Kyoto net position model

The *Kyoto Net Position model* provides annual emissions and removals associated with ARD activity since 1990. Removals from pre-1990 forests are not modelled, and all deforestation is assumed to be of pre-1990 forests harvested at age 28. While removals from post-1989 forests can be estimated beyond CP1, a constant rotation age of 28 is assumed. Harvesting and second rotation harvest residues are treated in the same way as in the UNFCCC inventory model.

3.7 External Review of the Current Approach

Greenhouse gas inventories submitted under the UNFCCC are subject to review by a team of experts nominated by Parties to the Climate Change Convention. New Zealand’s greenhouse gas inventory was reviewed in 2001 and 2002 as part of a pilot study of the technical review process, where the inventory was subject to detailed in-country, centralised and desk review procedures. The inventories submitted for the years 2001 and 2003 were reviewed during a centralised review process. The 2004 inventory was reviewed as part of the in-country Kyoto Protocol initial review held from 19 to 24 February 2007. (MFE 2008a). Review reports are available from the Climate Change Convention website (www.unfccc.int).

The latest expert review report (UNFCCC, 2007) concluded that:

- “New Zealand’s greenhouse gas inventory is consistent with the Revised 1996 IPCC Guidelines and the IPCC good practice guidance, and adheres to the reporting guidelines under Article 7 of the Kyoto Protocol”
- “New Zealand’s national system is prepared in accordance with the guidelines for national systems under Article 5, paragraph 1, of the Kyoto Protocol and reported in accordance with the guidelines for the preparation of the information required under Article 7 of the Kyoto Protocol”

The annual Kyoto balance of emissions units projections have also been subject to reviews (AEA Technology 2005, 2007a, 2007b). The overall finding of the 2005 AEA Technology (UK) review was that “the methodologies employed to project emissions and sinks across the different sectors [are] generally sound and reasonable in their approach”. AEA Technology noted the uncertainties inherent in all countries’ approaches to projecting future greenhouse gas emissions, and that it is “not uncommon” for projections to change on re-analysis. They further stated that for the LULUCF sector, “methodologies and input assumptions are reasonable and the resulting removal and emission projections are of a good standard”.

However, to meet good practice, a process of continual improvement in inventory reporting is required.

3.8 Summary: Current Data and Models

NEFD area data are generally good for pre-1990 planted forests but less so for post-1989 forests, and require independent validation. Both historic and projected activity data are adequate at a national level, but lack detailed resolution over time and space. Identification of the extent of over-planting is currently weak, as is the extent of emissions as a result of land clearance for plantation establishment, although the latter has less impact on net removals.

Although the use of NEFD yield tables represents a pragmatic basis for deriving carbon stock estimates in the absence of a national forest inventory, there is wide variation in standing volume across sites, regimes and species in New Zealand, and trends over time may not be captured adequately. Conversion of stand stem volumes to total stand carbon is well covered by C_Change, for radiata pine at least.

The combination of data and models has generally proved to be robust and useful for reporting carbon stock changes to meet international reporting requirements. However, the level of data accuracy is largely unknown, which is a key weakness to be addressed by LUCAS.

4. Alternative Information Sources or Assumptions

4.1 Introduction

LUCAS has been specifically designed to provide the information that will be used to report carbon stocks and stock changes in planted forests. However, the mapped areas that will define the activity data are not yet available, and only a small number of plots have been measured. This makes it difficult to ‘calibrate’ the current NEFD-based estimates to the estimates expected to be generated from LUCAS. International data are of limited use. IPCC default data are typically at a much coarser level of resolution, or are not relevant to New Zealand conditions.

4.2 Activity Data

4.2.1 Pre-1990 afforestation, harvesting and deforestation

Estimates have been made using a point-sampling approach, which suggests the planted forest area may be up to 20% higher than reported in the NEFD (Paul et al. 2007). However, the total area as at 1990 will come from LUCAS mapping, allowing annual afforestation estimates to be scaled. Most (94%) of the NEFD area has been classified by rotation number, which will provide an additional check on afforestation estimates and allow more accurate determination of historic age class distributions.

Pre-1990 afforestation probably involved a higher rate of over-planting than post-1989 planting. For example, 28 000 ha of natural forest conversion was identified between 1980 and 1988 in one source.¹⁵ The resulting emissions would affect the calculation of a 1990 decadal baseline.

4.2.2 Post-1989 afforestation, harvesting and deforestation

LUCAS is expected to provide estimates of afforestation and deforestation areas since 1990. LUCAS objectives include (MFE 2008a):

- determining changes in land use between 1990 and the start of the first commitment period by providing a New Zealand-wide map of land use at 1990 and at 2008
- determining changes in land use through the first commitment period by providing a New Zealand-wide map of land use at 2012
- determining where forests have been harvested, and where deforestation has occurred.

This mapping work will also provide estimates of the extent of “over-planting” of existing forest. While the quantitative analysis required to improve estimates of the area of forest over-planting has not been completed, it is suggested that a reasonable Most-likely assumption would be 10–12% (reduced from 16% used in the Net Position report), with 8% best Case, and 16% worst-case (Craig Trotter, Landcare Research, pers. comm.). The impact of the changed assumptions in CP1 would be an improvement in expected net removals by 2 Mt CO₂-e (Table 4).

¹⁵ Forestry Facts and Figures 1990 NZFOA Inc.

Table 4 Impact of over-planting assumptions

	2008 Net Position Report	Alternative 1
	Assumed over-planting proportion (%)	
Best case	8	8
Most likely	16	12
Worst case	21	16
	Impact of assumption on CP1 base net removals (Mt CO₂)	
Best case	-7.8	-7.8
Most likely	-15.6	-11.7
Worst case	-20.5	-15.6
Mean impact of over-planting*	-13.64	-11.34
Mean Net Planted Forest Uptake**	84.08	86.38

* From @Risk Monte Carlo analysis assuming triangular distribution

** Net uptake in planted forests during CP1 before deforestation emissions are subtracted.

Table 5 Potential future afforestation

Study	Potential area (ha)	Comments
Harris et al. (1979)	6 000 000	Reduced from reported 7 million ha to account for actual 1980–2007 afforestation.
Hale and Twomey (2006)	3 764 705 9 279 854	Low cropping suitability (NZLRI) Unsuitable for cropping (NZLRI) (NZLRI: NZ Land Resource Inventory)
MAF ¹⁶	1 179 900	Class VII and VIII land that should be taken out of production
Trotter et al. (2005)	1 450 000	Marginal pastoral farmland suitable for natural revegetation (from LCDB, NZLRI and LCDB databases)
MAF (2007)	200 000	East Coast scheme (37 000 ha planted)
Landcare Research (Giltrap et al. 2003)	278 870	Maori pastoral land suitable for forestry
Royal Society (2006)	587 000 2 525 000	North Island South Island Energy farming using <i>Salix</i> -excluded land over 1000 m in elevation, land over 15° in slope, land parcels less than 1 ha, DOC land and native forest, land already in plantations, dairying and horticulture land, and land returning over \$350/ha/year
Hall and Gifford (2008)	Minimum: 86 513 + 744 367 Maximum: 5 127 000	North Island + South Island Minimum: LCDB2 Low producing grassland (41), Depleted grassland (44), Gorse and broom (51), Mixed exotic shrubland (56), and Agribase farm types beef, sheep, deer, and minor unallocated categories (BEF, DEE, GRA, NOF, SHP, SNB, UNS). Excluded North Island > 800 m, South Island > 700 m; slope > 45°; LUC 1–IV and VIII; Conservation estate. Maximum: includes LUC IV, altitude < 1000 m, medium quality pasture.

Several studies have attempted to quantify the amount of land available for plantations, as summarised in Table 5 (above). The most recent and detailed analysis of marginal land availability is provided in Table 6, which follows the analysis methodology given in Sutherland et al. (2006), but extended in four ways:

- (i) The analysis includes privately owned, as well as Crown-owned, lands.

¹⁶ Dominion Post, March 27, 2007.

- (ii) A more conservative approach is taken to estimating available land area, by first removing areas recorded as woody vegetation in LUCAS Ecosat imagery from areas recorded as grassland in the LCDB2 database.
- (iii) An alternative definition of marginal land is used, that is more conservative (i.e. the resultant lands are more “marginal”) than the original definition—with analysis to the original definition retained for comparison. The analysis comprises all classes 7 and 8 land, plus class 6 lands with an erosion potential rating in the NZLRI database (Eyles 1985) of:
 - either moderate to extreme (ratings 2-5)—the original definition
 - or severe to extreme (ratings 3-5)—the more conservative scenario.
- (iv) The analysis provides a breakdown of land ownership area by (Regional Council) region.

Further details of the analysis methodology can be found in Chapter 4, Section 5.

Table 6 Marginal lands available for exotic forestation

For each region, two entries are shown: the upper includes Class 6 lands with erosion potential ratings of moderate to extreme, and the lower Class 6 lands erosion potential ratings of severe to extreme. Both entries include all Class 7 and 8 lands

Region	Crown Land (ha)	Private Lands (ha)	
		Total	Maori-owned
Northland	14 314	189 356	7808
	11 272	108 783	6809
Auckland	1896	46 239	140
	395	12 314	43
Bay of Plenty	4730	88 704	9741
	3776	21 954	2993
Waikato	18 109	320 314	13 388
	6550	106 006	5315
Taranaki	2120	90 850	388
	1217	30 063	108
Gisborne	4580	286 641	39 367
	2001	142 147	23 060
Manawatu-Wanganui	16 727	632 270	14 180
	8144	224 080	2766
Hawke's Bay	13 198	351 999	23 356
	5098	149 801	11 475
Wellington	3747	164 287	957
	2056	65 971	447
Nelson-Marlborough	18 807	33 112	46
	14 004	15 230	42
Westland	14 960	10 797	31
	11 100	7048	29
Canterbury	92 084	75 857	35
	45 771	20 956	13
Otago	29 085	46 644	65
	15 135	12 392	7
Southland	13 147	27 343	172
	6313	4676	130
Total	247 504	2 364 413	109 674
	132 832	921 421	53 237

The data in Tables 5 and 6 clearly indicate there is scope to expand the total planted forest area in New Zealand in a major way, with significant attendant environmental benefits. The scenarios that have been used to project future afforestation at a national level are within even the more conservative of the area limits suggested above for marginal lands (i.e. if forestry is restricted to just the most erosion-prone Class 6 lands). In terms of achieving the potential indicated, in the absence of incentives, it has been suggested there is a close relationship between the annual rate of planting and expected financial returns from forestry (Horgan 2007). Only a modest level of new planting was expected in the next decade on this basis. Other analysts have suggested that the Permanent Forest Sinks Initiative may reduce the amount of land conversion to plantations that would otherwise take place, unless equivalent incentives for carbon sequestration by plantations are provided (Hendy et al. 2006). Such incentives have been recently introduced under the Emissions Trading System for forests, and the Afforestation Grants Scheme, which aim to achieve planting of some 20 000 ha of new forests annually.

Future harvesting decisions have long been identified as a key driver for future net removals (Ford-Robertson et al. 2000). There are no “official” estimates of future harvesting—this is discussed further in the section on models.

Deforestation emission estimates need to be recalculated based on actual deforestation projects, with care taken to describe the areas involved and resulting stock changes correctly. Immediately before deforestation a forest may include stocked areas, temporarily and permanently unstocked areas, and riparian or reserved vegetation zones. After conversion to another land use (normally grazing), some of the stocked and reserved vegetation may be retained. The “before” and “after” stock estimates must take this into account.

A scenario-based approach to future afforestation and deforestation is ultimately necessary, supplemented with more detailed information about specific deforestation projects as this becomes available.

4.3 Stem volume yield tables

A key question is the relative average growth rates of pre-1990 and post-1989 forests. It has been suggested that using an area-weighted yield table based on the entire planted forest resource is likely to result in conservative estimates of CO₂ removals in CPI, because of a belief that the post-1989 resource is more productive (Wakelin et al. 2008b). The difficulty is in quantifying that productivity improvement. For example, an analysis of Scion PSP data was used to derive a post-1989 “Best-case” yield table based on a 300Index value of 27 (Wakelin et al. 2008b). An independent analysis carried out for the Kyoto Forest Owners Association assumed a 300Index value of 29. In the absence of a nationally representative sample, or more complete knowledge of the resource in terms of the area by location, species, regimes, ownership and management, it is difficult to justify any particular set of assumptions on which to base yield estimates.

4.3.1 Post-1989 resource characteristics

Our knowledge of the characteristics of the post-1989 resource has improved a little in recent years, as the NEFD survey now requests areas classified according to rotation number. It is therefore now possible to derive an area-weighted yield table based only on post-1989 first rotation stands. This allows the different post-1989 mix of species, region or regime to be reflected in the yield table.

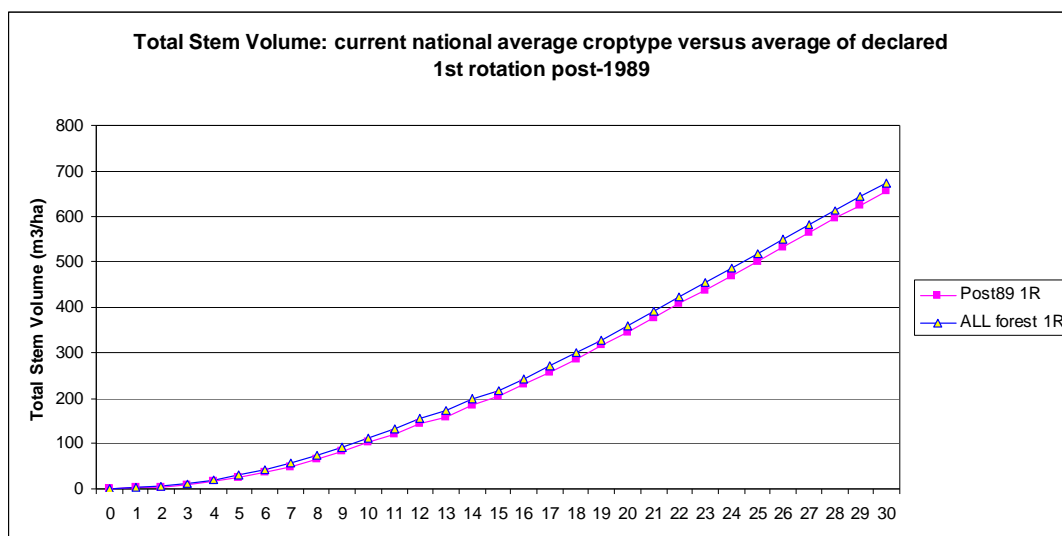
It does not reflect productivity differences, however, because the 1995 NEFD yield tables do not differentiate between pre- and post-1989 yields.¹⁷

Of the 1.8 million ha of planted forest in the 2006 NEFD database, 60% was classified as 1st rotation, 30% as 2nd rotation, and 10% remained unclassified. The post-1989 1st rotation resource is 84% radiata pine, compared with 89% for the resource as a whole. Differences in the proportion by region are more pronounced—post-1989 afforestation has been relatively evenly spread across eight of the ten regions, whereas the Central North Island had previously been dominant. There has also been an increase in the radiata pruned and waste thinned regime at the expense of the minimum tending regime.

However, differences in the species, region and regime proportions are too small in themselves to have a major impact on average yields. For this report, an area-weighted yield table was prepared from the post-1989 first rotation crop-types (Figure 1). It is slightly lower than the national average, but it is within 10 m³ or 10%, whichever is the greater, at all ages. The inclusion of any unclassified areas as first rotation would tend to increase yields.

This analysis needs to be repeated when the revised NEFD yield tables are complete. The key factors in the revised NEFD yield tables influencing the area-weighted post-1989 yield table are likely to be productivity assumptions for post-1989 forests, and the distribution among regions. Indications are that pre-1990 yields will be lower, while post-1990 yields will be at a similar level to the 1995 yield tables (Paul Lane, MAF, pers. comm.). However, it should be recognised that the NEFD yield tables have been prepared for wood availability forecasting rather than for carbon inventory purposes, so careful interpretation will be required. For example, the relationship between merchantable stem volume and total stem volume may differ significantly between regions depending on pulp markets. Until the set of revised NEFD yield tables is complete, current NEFD data generally support the yield tables used in the Net Position analysis.

Figure 1 Area-weighted average post-1989 1st rotation yield table compared with the UNFCCC inventory national area-weighted average yield table. Both are based on the same underlying yield tables, but differ in the proportions by species, regime and region



¹⁷ There are differences between pre- and post-1975 yields in some crop-types, but the area of pre-1975 radiata pine is now small and carries little weight in the current yield table.

4.3.2 IPCC defaults

Table 3A.1.7 in the *GPG-LULUCF* (“Average Annual Above Ground Net Increment in Volume in Plantations by Species”) gives a range for radiata pine *stem volume increment* of 11–35 m³ ha⁻¹ year⁻¹, with a mean of 23.5. The range has been taken from the summary appendix in Ugalde and

Prez (2001), which is sourced to two references on tropical and subtropical plantations. However, the text actually quotes a range of 11–50 for radiata pine in New Zealand. The upper limit can be referenced to Shula (1989). Clearly, individual stands can grow at very different rates, and the IPCC defaults can provide little guidance on a suitable national average growth rate in New Zealand plantations. The mean annual increment at maturity of the area-weighted NEFD yield table is about 20 m³/ha/year, which is comfortably within the range suggested.

Similarly, the IPCC defaults for whole tree *wood density* are of limited use, given that radiata pine wood density in New Zealand has been extensively studied and is known to vary with temperature, soil fertility, genetic stock and age. The variation in radiata pine outer-wood density at breast height is significant, ranging from 350–600 kg/m³, with the upper limit occurring on warm, low fertility sites (Beets et al. 2007). The effect of age on density is captured within the carbon yield tables used in the UNFCCC inventory, so the average density of harvested logs in the model varies with the average clear-fell age, and is higher than the average density for the growing stock, which also varies over time due to the uneven age class distribution. A single IPCC default density value would not reflect these differences.

4.3.3 Validation against LUCAS data

A limited number of plots are available from the first measurement season. These plots have been analysed and compared with the NEFD-based national average yield table. For each subplot, plot measurements were used to estimate 300Index and Site index values and current stand volume. These were used to simulate the growth of the subplot from establishment to age 30, using available regime information. The resulting volume increments were used as input to C_Change to predict stand carbon from establishment to age 30.

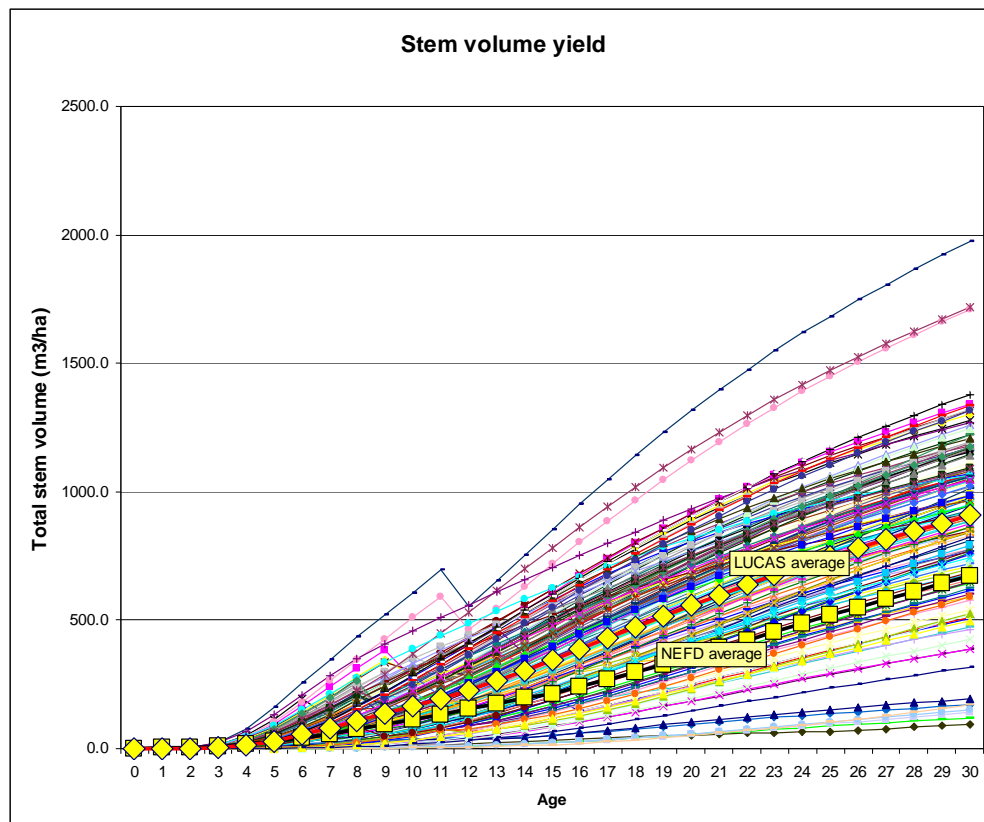
Figure 2 shows the stem volume yield tables generated for each subplot, together with a yield table derived from mean subplot values at each age (“LUCAS average”), and the national average NEFD-based yield table (“NEFD average”). This preliminary analysis suggested stem volumes could be some 50% higher than assumed by the NEFD yield tables. Further analysis of this data is underway, and it is not yet possible to say whether a nationally representative sample of plots will show a similar difference. Modelled density was lower in the LUCAS plots, meaning the overall difference in terms of stand carbon was reduced to about 30%. Both the higher volumes and lower densities are consistent with post-1989 afforestation being concentrated on ex-farm sites. However, differences in regime details modelled (particularly stocking) will also have influenced the result.

4.3.4 Other sources of stem volume estimates

The new NEFD yield tables provide a view of post-1989 productivity that appears to be at odds with LUCAS plots measured to date. Before they can be used, there will need to be careful analysis of the reasons for this difference (for example, the possible exclusion of pulp volume in regions where there is no pulp market). Similarly, the deforestation and afforestation yield tables prepared for use with the ETS have been produced for a specific purpose, and are not a useful substitute for LUCAS data or the current NEFD yield tables.

For future afforestation, GIS overlays of the variables required as input by the 300Index growth model and C_Change will allow projections to be made on the same basis as used within LUCAS. A model for predicting 300Index and Site Index has been developed using regression kriging (Palmer et al. 2008), with spatial datasets including climate, land-use, terrain, and environmental surfaces. This provides maps describing the spatial variability of potential *Pinus radiata* productivity across New Zealand, with a known level of certainty. If similar information was available for use with C_Change (e.g., variables affecting wood density), this would be a good basis for projecting future CO₂ uptake by new plantations, and by existing plantations where current age and past management is known. Remote-sensing could assist in providing sufficient missing information for existing stands to allow projections to be made (e.g., mean top height). Alternatively, NEFD age class information could be used as the basis for allocating age classes to planted forest GIS layers.

Figure 2 Stem volumes per hectare by age for LUCAS subplots (at age of measurement and as modelled by 300Index/C_Change), subplot averages (“LUCAS average”) and the NEFD-based national average yield table (“NEFD average”)



4.4 Carbon Pools

4.4.1 Above-ground biomass

A biomass conversion and expansion factor (BCEF_s) can be derived from the national average carbon yield table, for comparison with IPCC defaults (Figure 3). The IPCC default values are for temperate pines from Table 4.5 in the 2006 Guidelines (IPCC 2006), and vary with growing stock. The relationship in C_Change is consistent with the defaults.

4.4.2 Below-ground biomass

Below-ground biomass is derived from above-ground biomass using a root:shoot ratio. Figure 4 compares the IPCC default ratios for temperate conifers (Table 4.4 in IPCC 2006) with the ratio from the national average carbon yield table. More recent work on root:shoot ratios in New Zealand radiata pine recommends that a ratio of 0.2 be used across all stand ages and sites (Beets et al. 2007). This would result in a reduction in total stand carbon of about 1–2% (or more at very young ages).

4.4.3 Dead-wood and litter

Dead organic matter comprised about 20-25% of the total plantation carbon stock during the period from 1990 to 2018 (Steve Wakelin, unpublished calculations). Over this time frame, annual increases in the dead organic matter stock (mainly post-harvest) are in the range 1–2 kt C—sufficient to make dead organic matter a Key Category in the inventory. Reporting of Dead-wood and Litter in land remaining forest land is optional under Tier 1 analyses, but this is not appropriate for a Key Category. There are regional defaults for the litter component of the Litter pool, but not for fine woody debris or any part of the Dead-wood pool.

The approach to modelling harvesting may need to be revised given the importance of harvest residues in the inventory (see the following section on Models).

Figure 3 Comparison of New Zealand-specific and IPCC biomass expansion factors

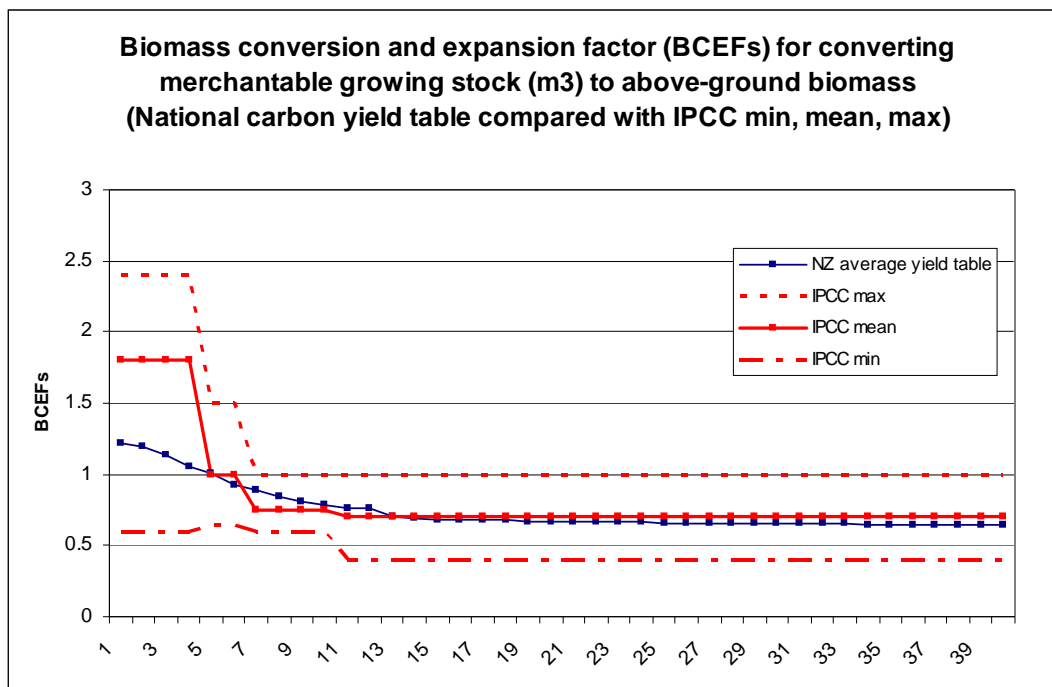
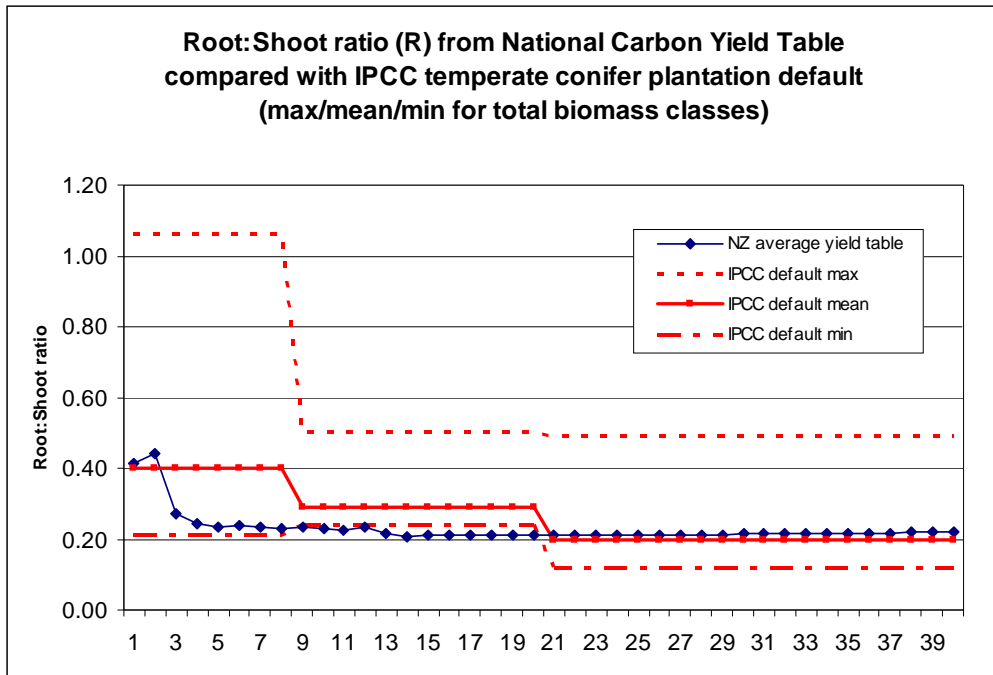


Figure 4 Comparison of New Zealand-specific and IPCC root:shoot ratios

4.4.4 Other species

There is ongoing work to calibrate C_Change for use with species other than radiata pine—particularly Douglas fir and eucalypts. This would account for 97% of the planted forest resource.

Table 7 Impact of soil C assumptions on CP1 net position

	2008 Net Position Report	Alternative 1	Alternative 2	Alternative 3	Alternative 4
Scenario	Assumed Soil Carbon change with Afforestation (t C per hectare)				
Best case	0	0	-4.7	0	-8
Most likely	-4.7	-9	-9	-12	-12
Worst Case	-18	-12	-12	-18	-16
	Impact of assumption on CP1 net removals (Mt CO₂)				
Best case	0	0	-2.9	0	-5.0
Most likely	-2.9	-5.6	-5.6	-7.5	-7.5
Worst Case	-11.2	-7.5	-7.5	-11.2	-10.0
Mean impact of soil C change¹	-4.95	-4.21	-5.31	-6.08	-7.49
Mean Net Planted Forest Uptake²	84.08	84.82	83.72	82.95	81.54

¹ From @Risk Monte Carlo analysis, assuming a triangular distribution.

² Net uptake in planted forests during CP1, before deforestation emissions are subtracted.

4.5 Soil Carbon

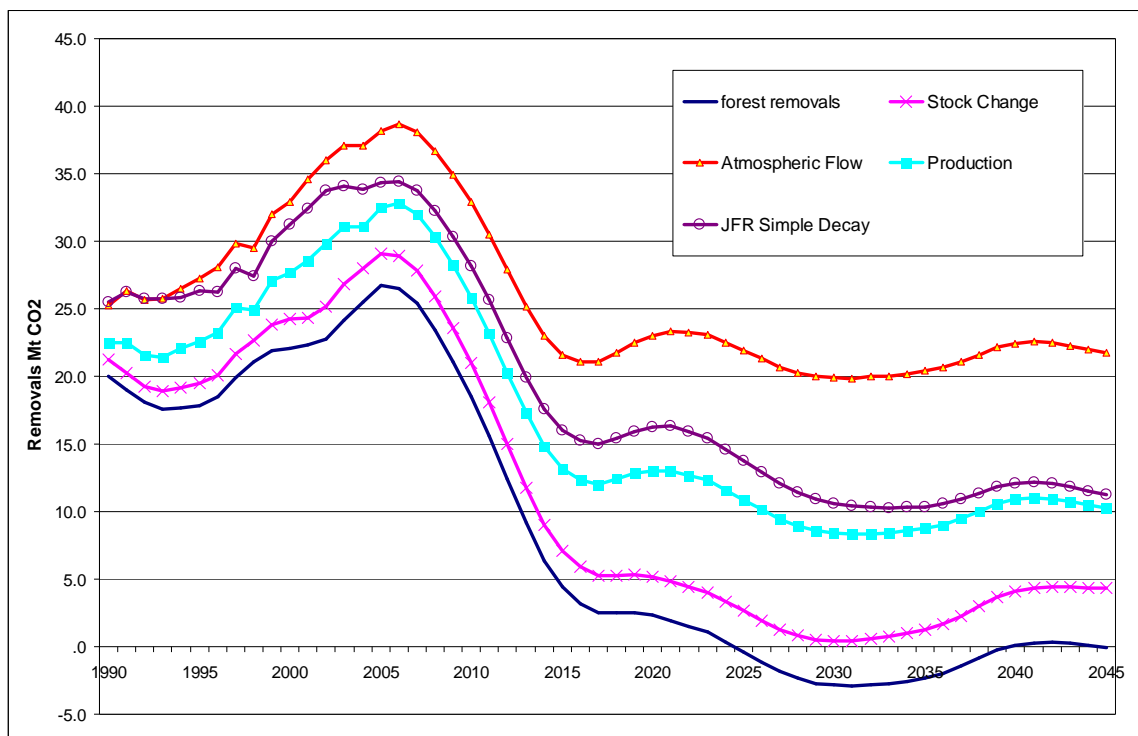
Re-analysis of the national soils data suggests that after removing auto-correlations, the best estimate from the Soils CMS is a reduction of soil carbon following afforestation of pasture of 12 t C/ha (rather than 18 t C/ha as used in the Net Position model (Craig Trotter, Landcare Research, pers. comm.)). While this brings the estimate closer to that obtained for radiata pine from the paired plot database (-9 t C/ha), the impact of these changes in CP1 is not great under the present assumptions used for calculations, as shown in Table 7 (above). However, the present methodology is based on the key assumption that change in soil carbon is represented by a triangular distribution comprising worst-case, most likely and best-case values. If mean values of current datasets are used, which implies that the means are unbiased despite their large standard deviations, the changes become more significant. These issues are discussed further in Chapter 5.

4.6 Models

4.6.1 Harvested wood products

Stem carbon removed offsite at the time of harvesting is currently treated as an instant emission, according to the IPCC default methodology. Four alternative approaches have been put forward for accounting for the harvested wood products pool. These indicate that the harvested wood products pool is growing rather than being static, so all approaches result in an increase in net removals from forests compared with the IPCC default approach (Wakelin et al. 2008a). The magnitude of the contribution from harvested wood products to net forest removals varies, as illustrated in Figure 5.

Figure 5 Impact of harvested wood products accounting approaches on net forest removals (all planted forests). Positive values indicate net removals; negative values net emissions. Source: Wakelin et al. 2008a



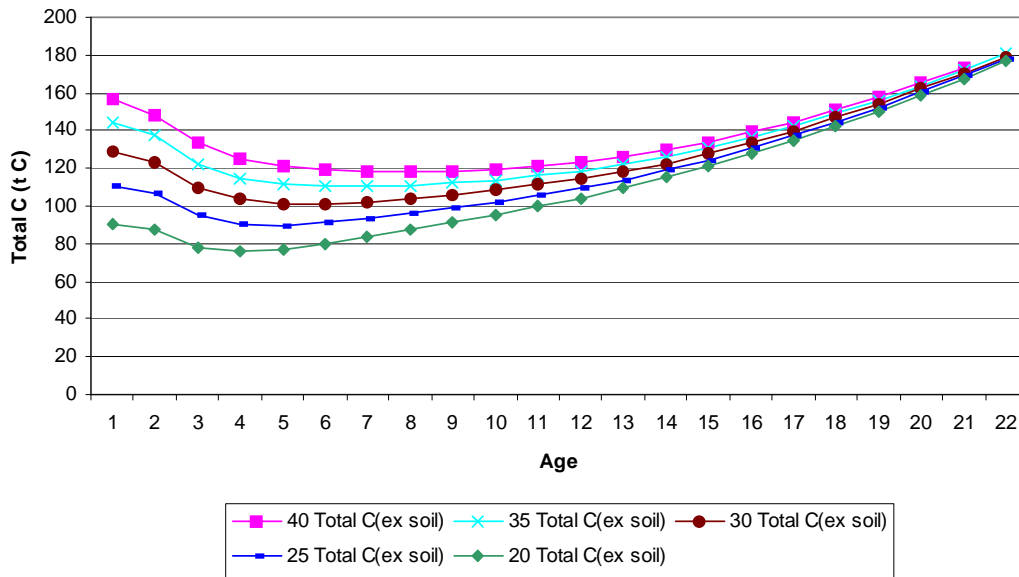
A hybrid between the Stock Change and Production approaches has also been suggested as an option for post-2012 accounting, because it would remove the need to account for exports (where the type of use is unknown and difficult to influence) and imports (where the sustainability of production may be unknown) (Schlamadinger et al. 2007b). Such an approach would minimise the contribution of harvested wood products to net removals for a wood exporting country like New Zealand. Another option is to apply a Simple Decay approach to harvested carbon as some parties are already doing in their inventories. This would meet the requirement to improve the accuracy of inventory emission estimates. While harvested wood product accounting is not considered further in this report, it clearly has the potential to be a significant factor in post-2012 accounting.

4.6.2 Harvest residues

The current models initialise harvest residues at the start of second rotation stands using the total stand carbon at age 28 in the national average radiata pine yield table, less the proportion of stem carbon assumed to be removed as harvested wood products. One problem with this approach is that initial harvest residues do not change to reflect the characteristics of harvested stands. For example, as rotation lengths are extended, the stand carbon present at harvest and the initial level of post-harvest residues should both increase. However, in the models stand carbon increases at harvest, but post-harvest residues remain constant. The extra quantity of residues that would be expected becomes an instant emission.

Figure 6 compares second rotation yield tables following five different first rotation clear-fell ages. The length of time for second rotation yields to surpass the initial level of residues increases with previous rotation age. Harvest residues could be calculated correctly by using a unique second rotation yield table for each clear-fell age. A simpler alternative may be to remove harvest residues from the yield tables and calculate their decay externally based on the stand carbon present at clear-fell.

Figure 6 Total stand carbon per hectare (excluding soil) for second rotation carbon yield tables assuming different previous rotation harvest ages



A series of models was run to examine the impact on stocks and net removals of alternative treatment of harvest residues. These models were run as variations of the post-1989 afforestation model with a 28-year rotation, but with:

- (i) all post-harvest residues assumed to be collected and burned completely for bio-energy (that is, instant emissions of harvest residues)
- (ii) 50% of post-harvest residues collected and burned completely for bioenergy; remaining 50% stays on site and decays
- (iii) all post-harvest residues collected and converted to biochar. The conversion assumes that 50% of available carbon would be converted to biochar by the pyrolysis process, with the remainder instantly emitted. Biochar is assumed to remain undecayed, with no other soil carbon changes
- (iv) 50% of post-harvest residues collected and converted to biochar. Uncollected residues decay on-site, while the biochar conversion is as for model (c).

Scenarios (i) and (iii) represent extreme situations with all post-harvest residues collected. In reality, while collecting all post-harvest residues may be technically feasible, it involves considerable cost. Scenario (ii) and (iii) both assume only 50% of residues would be collected, which may still be an over-estimate given that the resource contains a high proportion of steep and/or remote sites. Post-1989 forest carbon stock and net removals are compared with the Base scenario (all post-harvest residues decay in the forest) in Figures 7 and 8.

During the period of concentrated harvesting when net removals are negative, the 50% biochar and biofuel scenarios result in net removals that are 5–10 Mt CO₂ per year lower than the base scenario. Between cycles the scenarios increase net removals by up to 5 Mt CO₂ per year.

Figure 7 Carbon stocks under alternative harvest residue management scenarios

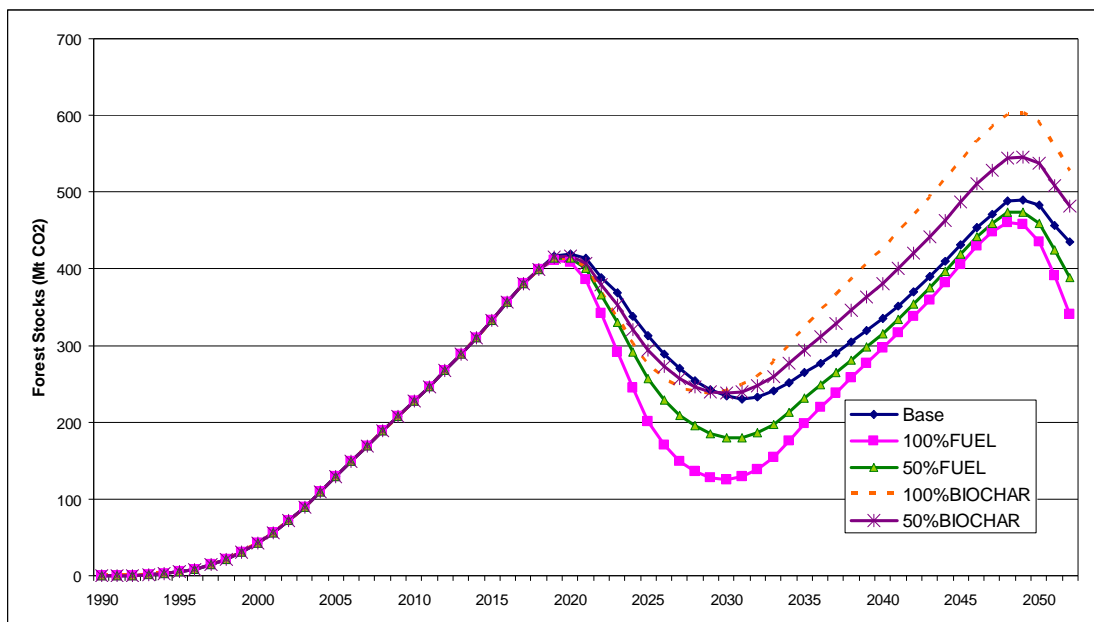
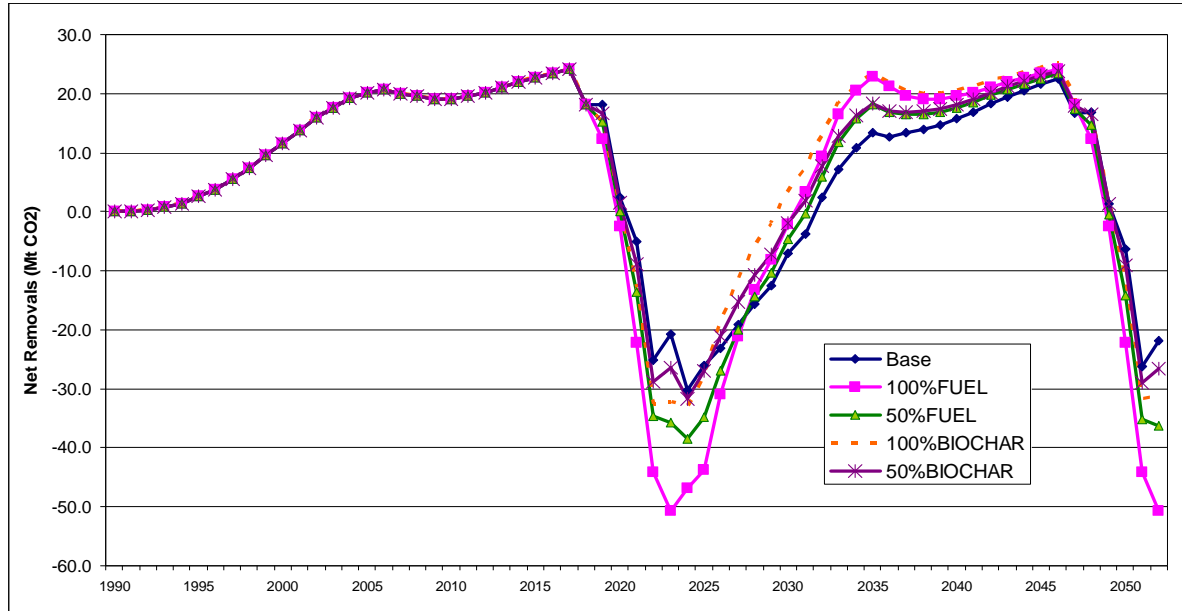


Figure 8 Net carbon removals under alternative harvest residue management scenarios Positive values indicate net removals; negative values net emissions



4.6.3 Harvest Yield Regulation

Future projections of net removals are driven by the current age class distribution, future afforestation and the timing of harvesting. The latter can be modelled using a fixed rotation age, or harvest yield regulation constraints that place limits on the annual harvest. The UNFCCC inventory models include constraints that require the annual harvest to be non-declining, and the maximum increase in harvest between any successive years to be less than 10%. These constraints have the effect of smoothing out peaks in harvesting that would otherwise occur due to the uneven age class distribution, and therefore smoothing out fluctuations in net removals.

The model does not differentiate between pre-1990 and post-1989 stands when harvesting decisions are made, nor does it distinguish between stands owned by large scale owners and those of small growers – unlike MAF’s wood availability forecasts (MAF 2006).

Variations in harvest yield regulation have the potential to affect the timing and magnitude of fluctuations in net removals. Options could include replicating the harvest volumes from NEFD wood availability forecasting. These are based in part on company intentions, but would give questionable results unless the same yield tables were used. Harvesting could instead be determined using an economic model— an approach that was used within the HWP model, based on the Global Forest Products Model. Alternative yield regulation approaches are reported for pre-1990 forests in the results.

4.6.4 Deforestation modelling

The simple deforestation model can be improved by attempting to model the main large-scale deforestation projects that have been undertaken, including attempting to capture the actual pre-harvest carbon stocks and their fate under the conversion approaches adopted. Analysis is currently underway.

4.6.5 UNFCCC inventory model

There is scope to increase the resolution of the current models by retaining crop-type-level yield tables, and to use the information on rotation number now available within the NEFD to more accurately attribute post-harvest residues. Proposals for improving the model are under consideration, including the explicit modelling of historic deforestation.

4.7 Summary: Alternative Information Sources and Assumptions

Wall-to-wall mapping will ultimately provide estimates of land-use change and total plantation area, although not at a detailed age class level. The data could be used to calibrate the pre-1990 NEFD age class distribution and post-1989 afforestation estimates. It should also improve estimates of over-planting and emissions from site preparation for afforestation.

The NEFD volumes will be replaced by volumes obtained from LUCAS plots, with known sampling error limits. Wood density estimates should be improved using site data. New NEFD yield tables could be used to adjust current yields prior to LUCAS data becoming available (e.g. for the pre-1990 resource), but this will require a careful examination of the process used to derive the yield tables.

Several improvements to the models used to estimate carbon stocks can be made, particularly with respect to deforestation. More attention will need to be paid to the range of future harvesting scenarios and treatment of harvest residues. Alternatives to the instant oxidation assumption for harvested wood need to be further explored.

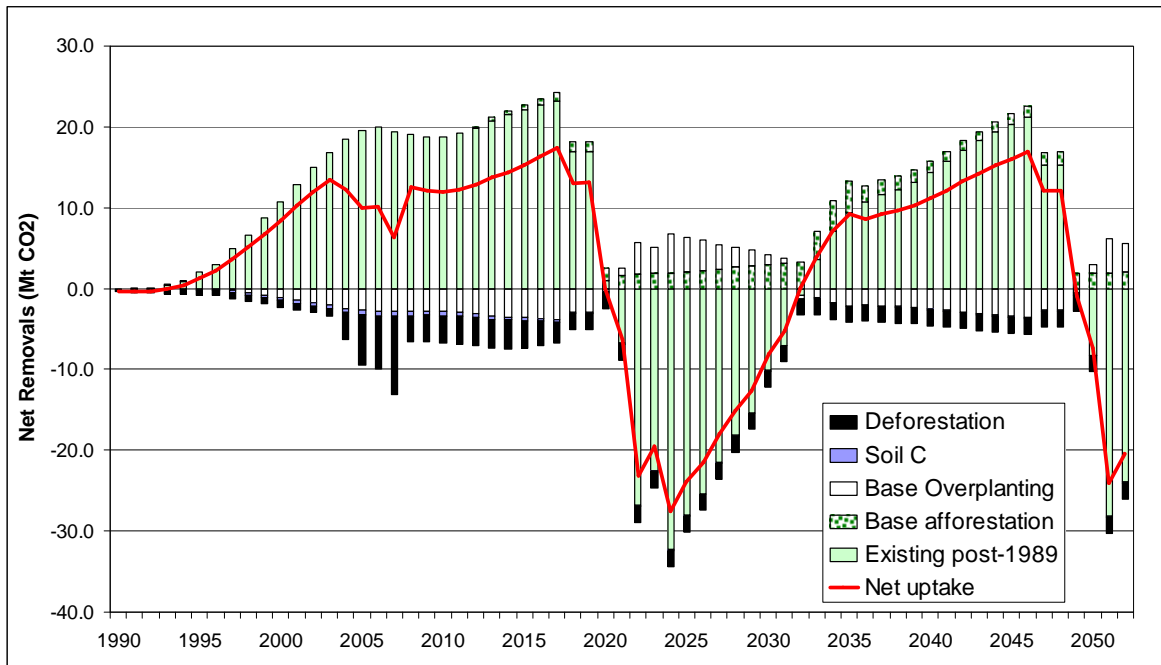
5. Current best estimates of planted forest carbon stocks and change

5.1 Stocks and Change under Current Land Use/Management (1990–2050)

5.1.1 Post-1990 planted forests

Net removals from by post-1989 forests were calculated for the Kyoto Net Position report, as shown in Figure 9. Net uptake is calculated from uptake from existing post-1989 forest (“Existing KF”) *plus* uptake from an assumed level of post-2007 afforestation (“Base afforestation”—5000 ha per year) *minus* ineligible uptake from over-planted forests (“Base over-planting”) *minus* soil carbon change (“Soil C”) *minus* deforestation emissions (“Deforestation”).

Figure 9 Net removals due to post-1989 afforestation and deforestation. Positive values indicate net removals; negative values net emissions



5.1.2 Pre-1990 planted forests

Net removals from pre-1990 forests are illustrated in Figure 10 from 1980 to 2050. These estimates are taken from the UNFCCC inventory model. The cyclical trend is due to the uneven age class distribution. The forest is initially immature, and net removals are positive as the average age increases. When large areas become available for harvesting, average maturity declines and the forest becomes a net source. The process then repeats.

The choice of 1990 as a baseline for net:net accounting is very unfortunate for New Zealand, as removals from planted forests were close to a peak in this year. Estimates of 1990 baseline removals are presented in Table 8. The use of averaging over a wider period only provides a small benefit in reducing the baseline. Note that these figures are probably under-estimated. The estimates were taken from the UNFCCC model results which derive 1980–1995 age class distributions from the 2006 NEFD age class distribution, using assumptions about historic harvesting, restocking and new planting. This process is used because the NEFD survey data for years before 1993 is incomplete, and the year-to-year variation in NEFD databases is large.

It is apparent from Figure 10 that the high baseline levels of net removals from pre-1990 forests are never reached again—this is because the area of pre-1990 forests is fixed, and over time the harvesting constraints smooth out the age class distribution. Over a longer time period the pre-1990 estate would probably become a normal forest, with no net gain or loss of carbon.

Figure 10 Net removals from pre-1990 planted forests. Positive values indicate net removals; negative values net emissions

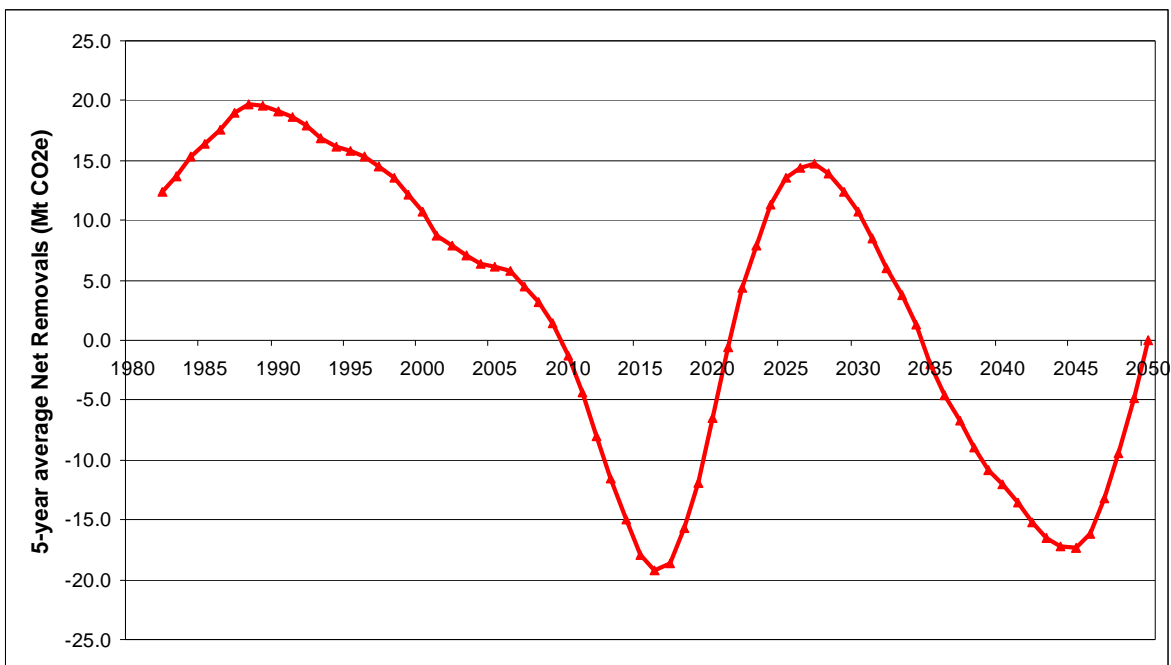


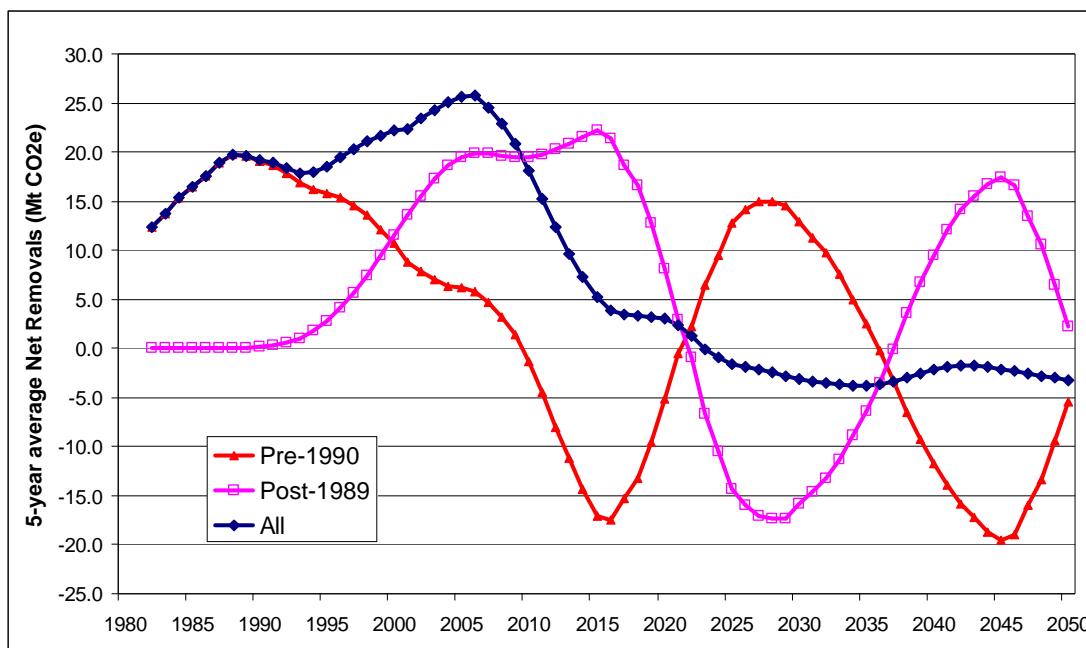
Table 8 1990 removals under alternative baseline assumptions

Baseline	CO ₂ removals (Mt CO ₂)
1990 removals	18.74
Average removals 1988–1992	18.70
Average removals 1985–1995	18.04

5.1.3 All planted forests

Figure 11 shows the estimates from all forests (i.e. pre-1990 forests and post-1989 forests, assuming no afforestation beyond 2007). Estimates are taken from the UFCCC inventory model.

Figure 11 Net removals for pre-1990 forests, post-1989 forests and all planted forests combined



Net removal trends in pre-1990 and post-1989 forests are counter-cyclical because of their respective age class structures—as post-1989 forests become available for harvesting, less harvesting is carried out in pre-1990 forests, allowing carbon stocks to build up again as that resource matures. The combined effect is for net removals in all planted forests to approach zero over time if the total area and rotation lengths are held constant. The effect of including pre-1990 forests in CP1 accounting is shown in Table 9. The overall reduction in CP1 net removals, after allowing for deforestation and soil carbon changes, would be about 15%.

Table 9 Removals (Mt CO₂) over the first Commitment Period from all planted forests (before deforestation emissions)

Forest	Scenario	CP1
Pre-1990 forests	Target age 28, non-declining yield	-23.8
Post-1989 forests	Fixed rotation age 28	+81.1
Post-1989 over-planted forests	Fixed rotation age 28	+14.6
<i>All planted forests</i>		+71.9

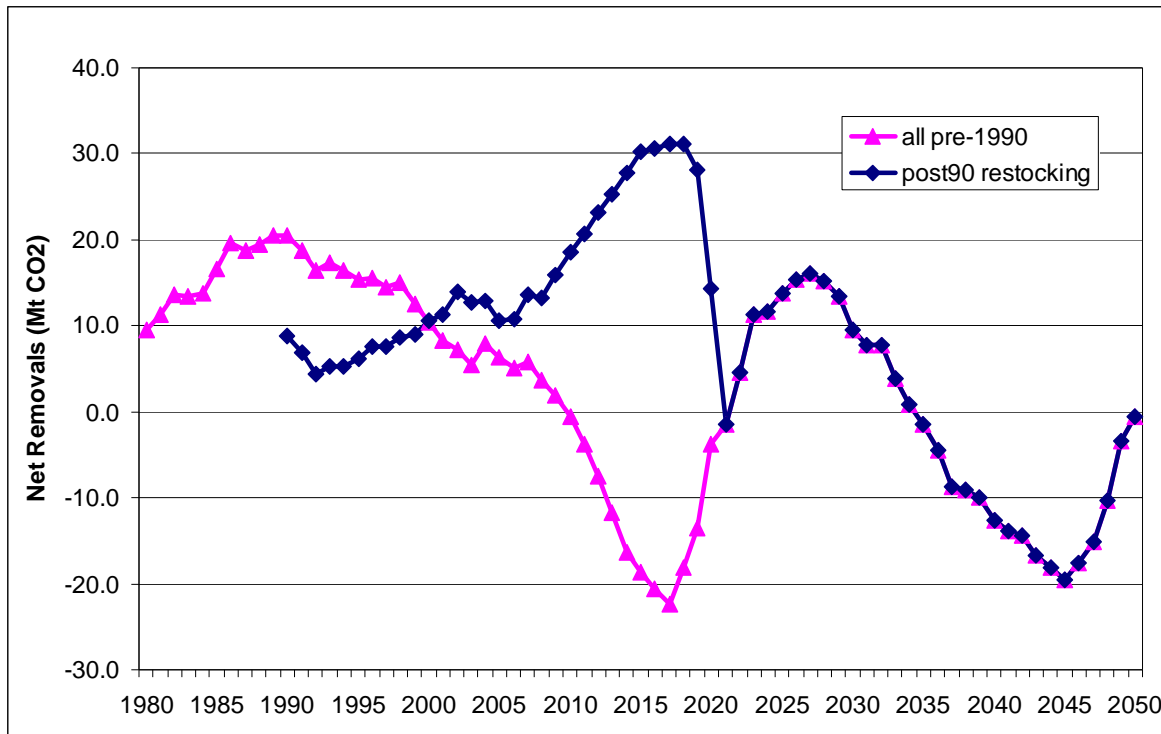
There are a number of possible ways in which pre-1990 planted forest may come into a post-2012 accounting regime. One is through the *Forest Management* provisions currently covered by Article 3.4, which allows carbon stock changes and non-CO₂ emissions on areas subject to forest management since 1990 to contribute towards the balance of units. This applies gross:net

accounting but (currently) with a cap on the total contribution from *Forest Management* as a surrogate for factoring out removals resulting from:

- elevated carbon dioxide concentrations above their pre-industrial level
- indirect nitrogen deposition
- the dynamic effects of age structure resulting from activities and practices before 1990 (Schlamadinger et al. 2007a).

Without a cap it is expected New Zealand would be required to factor out the impacts of anything that does not result from direct human action since 1990. This could be done through the use of activity response curves, baseline future scenarios, or an average carbon stocks approach (Canadell et al. 2007). One alternative would be to account for removals associated with post-1990 **restocking** of pre-1990 forests, on the grounds that such restocking is clearly human action (Fig. 12). Presumably the same argument could apply to emissions from harvesting of pre-1990 forests.

Figure 12 Net removals for all pre-1990 forests, and post-1990 restocking of pre-1990 forests.



Under an all-lands, net-net anthropogenic accounting approach referenced to 1990, there would be a need to factor out “age class effects”. If this is not done, then planted forests would be a liability in the accounts even with full replanting of harvested stands, simply because the level of removals in 1990 was so high. Using a 10-year baseline is useful for avoiding inter-annual variability captured in inventories (though not captured in the model reported here), but is still too short to reflect the dynamic nature of highly productive planted forests. There is provision in the *2006-Guidelines* to change the baseline if it was unusually different, and such a case could be made here.

6. Mitigation opportunities using planted forests

6.1 Afforestation

While afforestation can no longer contribute significantly to CPI net removals, it remains an obvious mitigation option for later CPs. Figure 13 (below) shows how afforestation of about 15 000 ha per year can prevent planted forests from becoming a net source.

As with the existing resource, any new forests may become a net source for periods in the future, but there will be an overall benefit seen by the atmosphere equivalent to the difference between the long term average carbon stock in the forested areas and that of the pre-afforestation land use.

If a constant annual area is planted for a period equivalent to the rotation length, a ‘normal’ forest will have been created, in which case net removals will be positive up until the year harvesting commences. After this point, harvest emissions will balance removals due to growth, unless afforestation continues.

Figure 13 Five-year average net removals from all planted forests under alternative post-2007 afforestation rates. [Positive values are net removals; negative values are net emissions].

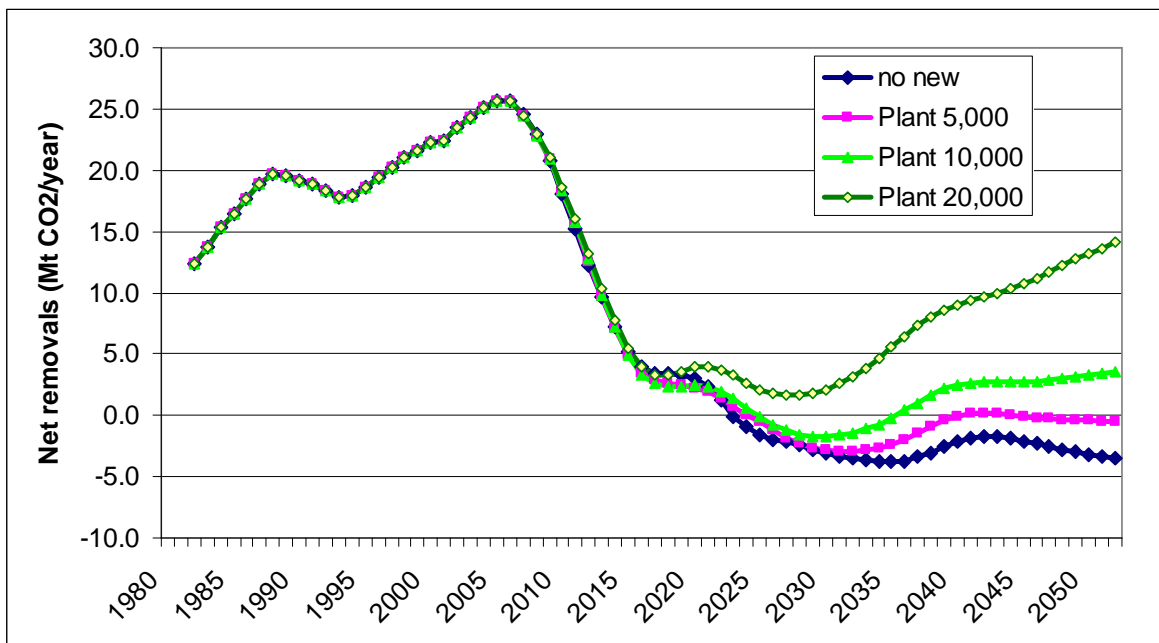


Table 10. Net uptake (Mt CO₂) by 5-year CP for alternative afforestation rates

	Annual afforestation rate 2007–2052 (ha/year)					
	0	5000	10 000	20 000	40 000	80 000
CP1	0	0.49	0.97	1.94	3.89	7.77
CP2	0	3.13	6.25	12.50	25.01	50.01
CP3	0	7.37	14.73	29.47	58.93	117.86
CP4	0	10.60	21.19	42.39	84.77	169.55
CP5	0	14.87	29.74	59.49	118.98	237.95
CP6	0	15.01	30.01	60.03	120.06	240.12
CP7	0	6.82	13.64	27.27	54.55	109.09
CP8	0	6.71	13.42	26.85	53.69	107.39
CP9	0	9.45	18.90	37.80	75.59	151.18
Total area Planted (ha)	0	225 000	450 000	900 000	1 800 000	3 600 000

Table 10 (above) shows net uptake (Mt CO₂-e) by five-year CPs for different post-2007 afforestation rates. Net uptake by forests established before 2007 are excluded. The national average yield table is assumed and land afforested is assumed to have no carbon stock present before planting. Uptake continues to be positive because planting continues past the establishment of a normal forest. A rate of 20 000 ha per year could be sustained from marginal farmland alone (total area planted over a 45-year period = 900 000 ha), while a rate of 80 000 ha per year would require conversion of better quality pasture as well.

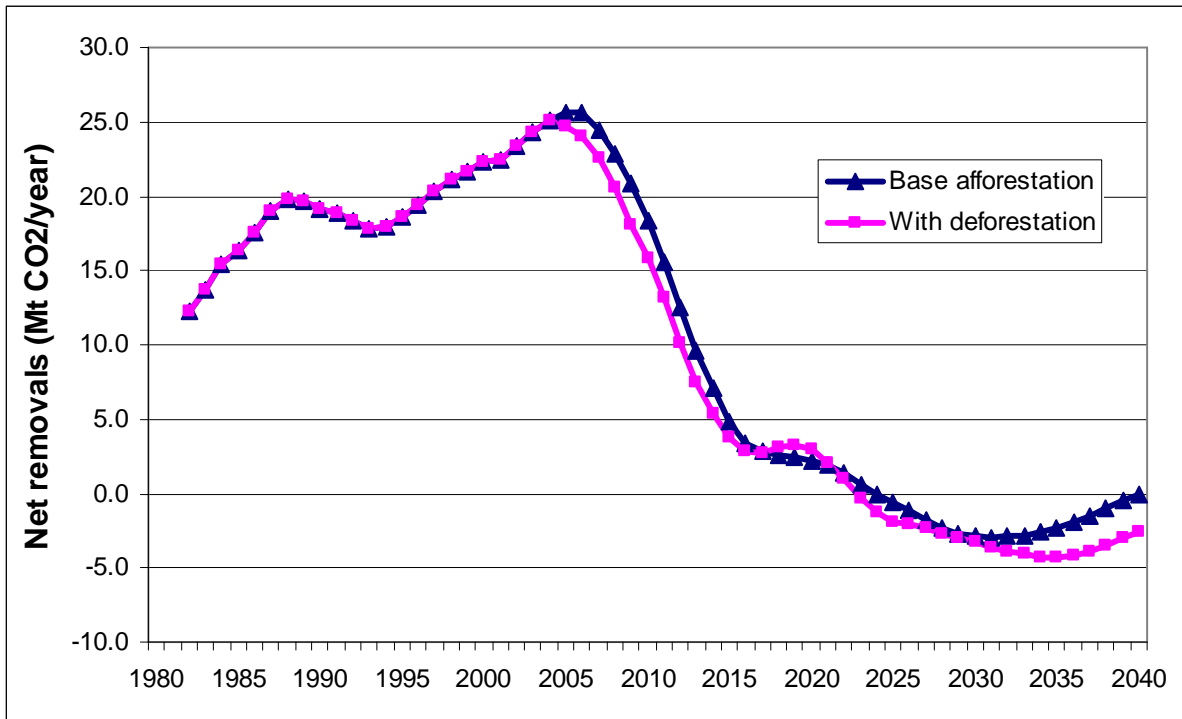
6.2 Deforestation

Reducing deforestation is another obvious mitigation strategy, given the importance of deforestation emissions in New Zealand's Kyoto accounts. Figure 14 shows the impact of deforestation on the base UNFCCC inventory model, which assumes an afforestation rate of 5000 ha per year. Deforestation areas were taken from the "Base scenario" deforestation intentions in Manley (2006) (Table 11). It was assumed that all carbon on deforested sites would be instantly emitted.

Table 11 Annual deforestation (hectares) for the deforestation scenario

Calendar Year	Area deforested (ha)
2006	12 700
2007	12 800
2008	11 000
2009	10 400
2010	11 000
2011	9600
2012	8000
2013	7100
2014	6200
2015	6200
2016	6000
2017	5200
2018	5100
2019	5100
2020	4900
2021–2065	5000

Figure 14 Impact of deforestation on net removals (all forests, base afforestation of 5000 ha/year). Positive values indicate net removals; negative values net emissions



The latest Net Position calculations assume that the ETS provisions will be successful in halting the conversion of plantations in most cases. There are also opportunities to use Kyoto accounting rules to minimise the impact of deforestation, for example:

- New Zealand argues that deforestation should be included with harvesting under the “A/R Debit rule” (also known as the “fast-growing forest fix” rule) and to have this rule apply in subsequent CPs. Restricting the rule to harvesting only is not consistent with the aims of the rule.
- New Zealand is also advocating a “flexible land use” approach, that would allow forest offset substitution—that is, deforestation emissions would not accrue if a forest was converted to pasture as long as an equivalent area of new forest is established elsewhere. There would be environmental benefits from such a change in approach, as it is likely to result in forests moving from less to more erodible land.

6.3 Forest Management

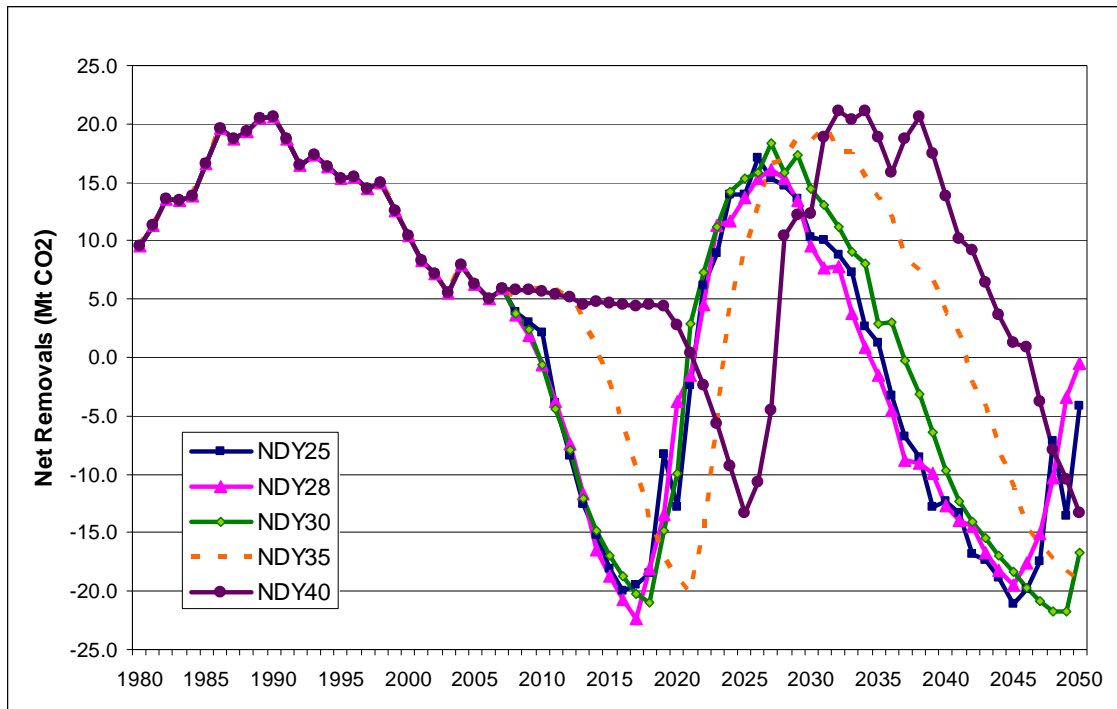
There are opportunities to increase carbon uptake in both the pre-1990 and 1990-2007 forest estates. This applies to the current crop and replanted stands. Forest management activities are also the subject of a separate report (Turner et al. 2008).

The main opportunity lies in increasing rotation lengths, and hence the carbon stock. Other options include:

- Increasing stand volume, e.g., through fertilisation, weed control, higher GF-rated seedlings and particularly through increased stand stocking. Not that there is also an interaction with wood density to consider.
- Increasing wood density. Wood density variation is reasonably well understood, allowing decisions on the siting of new planted forests to take this into account. For example:
 - Temperature variation from 8 to 16 degrees C, gives a range in density from 360–440 kg m⁻³
 - Fertility variation from high to low fertility, gives a range in density from 380–412 kg m⁻³
 - Density can also be manipulated in existing planted forests to a small extent by varying stocking, and to a larger extent through species or genotype selection for replanting. However, whether this increases total biomass at a given age remains an open question.
- Decreasing pruning—pruning results in a small transfer of above-ground carbon to the dead organic matter pools, from where it decays, and also suppresses tree growth. Pruned regimes also require lower stockings to promote diameter growth.
- Species selection—Some eucalyptus species may sequester more carbon than radiata pine in the short term. Other species (e.g., Douglas fir) may be able to hold greater carbon stocks in the longer term, despite being slower to build up the stocks.
- Biochar production—realistic assumptions for biochar production have yet to be developed, but the potential has been illustrated in the previous section on models.

Figure 15 illustrates the impact of varying target rotation age on net removals in the pre-1990 planted forest. When clear-fell ages are lengthened to 35 or 40, the forest remains a net sink for longer, as harvesting cannot increase until stands have matured. The cycle is then delayed, and higher peaks of net removals are reached.

Figure 15 Impact of target rotation age on net removals (e.g., NDY25 = non-declining yield with target rotation age 25)

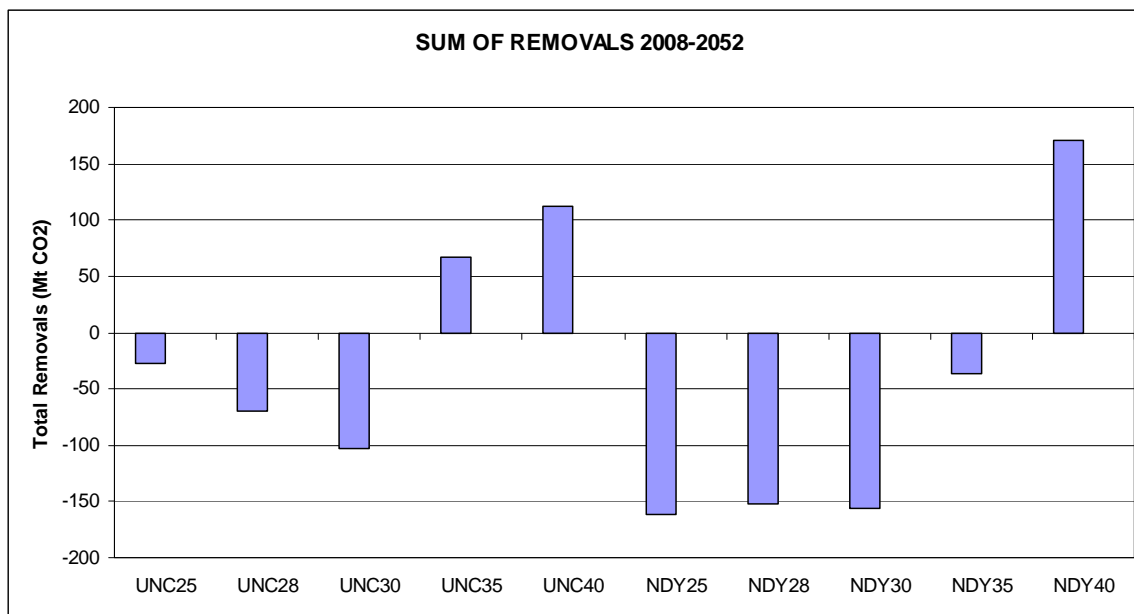


The impact of rotation age is related to the harvest yield regulation issues discussed earlier. Net removals in successive CPs are shown in Table 12 for two groups of scenarios with either unconstrained yields or non-declining yield constraints. Within each group there are five target clearfell ages, ranging from 25 to 40. Total removals from 2008–2050 are shown in Figure 16, with a trend towards higher removals with longer rotation ages. Total removals are influenced by rotation length, in that more or fewer rotations will be possible during the period examined.

Table 12 Net CO₂ removals by pre-1990 forests by Commitment Period (CP; Mt CO₂) for scenarios which differ in yield regulation (unconstrained versus non-declining yield) and rotation length (25 to 40 years).

CP		Unconstrained yield models by rotation length					Non-declining yield models by rotation length				
Starts	No.	25	28	30	35	40	25	28	30	35	40
2008	CP1	-112.6	-78.9	-38.2	77.7	137.9	-21.6	-23.8	-23.7	23.0	26.6
2013	CP2	-0.9	-42.8	-70.3	-28.4	96.8	-82.8	-86.5	-84.4	-32.2	22.5
2018	CP3	55.6	41.2	14.2	-88.7	-56.2	-10.4	-6.5	-5.1	-65.2	-3.7
2023	CP4	38.1	44.0	47.4	6.2	-120.2	70.0	68.0	76.5	51.8	-19.6
2028	CP5	9.5	27.3	20.4	31.0	-13.5	48.2	41.3	63.2	89.5	80.8
2033	CP6	-97.8	-15.9	17.3	-10.4	6.9	-14.1	-21.1	9.4	57.4	93.0
2038	CP7	-14.2	-71.8	-19.6	5.6	-42.4	-71.1	-68.1	-56.2	3.7	56.5
2043	CP8	54.5	-21.8	-70.4	76.8	-16.5	-82.8	-73.3	-95.2	-64.7	-6.2
2048	CP9	40.3	48.3	-4.1	-2.9	119.5	3.9	17.3	-40.2	-100.5	-79.2
	SUM	-27.5	-70.3	-103.4	66.9	112.2	-160.8	-152.7	-155.7	-37.1	170.6

Figure 16 Total net removals from 2008-2052 for yield regulation and target rotation age scenarios (e.g., UNC25 denotes unconstrained yields, rotation length 25 years; NDY40 denotes non-declining yield constraint, target rotation length 40 years)



There are other options related more to the accounting rules themselves, for example:

- Stands planted from 1990-2007 are only credited for growth during CP1 (2008–2012). There may be potential to maximise uptake in subsequent rotations by ensuring post-harvest residues are reduced to zero at the start of the second rotation. The additional emissions that would result are capped by the A/R debit rule, but full credits can then be earned on subsequent growth. Interaction between residues and soil carbon may nullify any gains, however, and the qualifying area is relatively small.
- The existing “D/R Loophole” effectively allows replanting of ‘deforested’ pre-1990 areas to count towards Kyoto targets, because all carbon stock changes (positive and negative) are tracked on forest land converted to other lands. This would potentially bring pre-1990 forest into a post-1989 accounting frame, after a ‘stand-down’ period in another land use. Reviewers may question this practice, however.
- HWP accounting would potentially reduce or delay emissions and possibly further influence forest management decisions, as factors such as species, density and wood age affect product use and life-spans.

6.4 Risk Management and Factoring Out

Previous estimates have not included adjustments for catastrophic damage caused by windthrow, fires, volcanic activity. Some post-2012 accounting options would make it possible to ‘factor out’ such events (Canadell et al. 2007). Risks from these sources are relatively low in New Zealand, particularly compared with fire losses in Australia, Canada, and the USA. Wind damage is more common, but can usually be subsumed within the normal harvesting activity. If emissions from these sources must be accounted for, then there is an opportunity to minimise risk through forest management, including the appropriate siting of planted forests, species selection, thinning regimes and the normal forest protection policies in place.

The likely impacts of climate change on New Zealand are summarised in the IPCC Working Group II report on Climate Change Impacts, Adaptation and Vulnerability, prepared as part of the Fourth Assessment Report (IPCC 2007) and by Mullan et al. (2008). Impacts on New Zealand include:

- reduced seasonal snow cover and rising snow line, potentially making more of the South Island high country available for afforestation
- water security problems in Northland, the east of the North Island and the north of the South Island
- significantly increased rainfall in the rest of New Zealand, causing flooding, landslides and erosion
- initial benefits to agriculture and forestry in western and southern parts of New Zealand and close to main rivers, due to longer growing season, less frost and higher rainfall
- reduced production in eastern parts of New Zealand due to droughts and fires, and the warmer and wetter weather could increase the frequency of upper mid-crown yellowing and winter fungal diseases.

The impacts on planted forests therefore include:

- the potential for the current resource to suffer catastrophic damage (e.g., from cyclones) before reaching maturity
- altered growth rates of the current resource—likely to increase in the south and west, but decrease in the east
- the introduction of new pests and diseases, or spread of existing ones in response to more favourable conditions
- the ability to compete with other land uses, including the expected impacts on agricultural production and economics, and the impacts of climate change on other wood producing regions (e.g., fire and drought are likely to negatively affect plantations in Australia's south and east).
- changes to the regulatory environment, e.g. in response to increased storm-induced erosion, or lower catchment water yields.

The expected impact of climate change on planted forests is still largely unknown, and is being addressed in a separate report (Watt et al. 2008).

6.5 Environmental Co-benefits and Risks

The environmental and social benefits of planted forests over-and-above their value for carbon sequestration are well recognised (Maclaren 1996; Clinton et al. 2006) and are essentially the same as for natural forests (Dyck 2003). Examples include:

- Soil conservation
- Increase in available woody habitat and biodiversity (native plants and animals)
- Recreation
- Landscape amenity
- Improved water quality; prevention of N and P leaching.

However, there are also negative impacts, whether real or perceived:

- Erosion associated with harvested sites
- Reduced water yields
- Increased animal pest populations
- Reduction in rural community services associated with declining population
- Reduction in landscape values

- Road congestion and noise from logging trucks.

These issues may become more severe over time in some places. For example, ECAN (the Canterbury Regional Council) already restricts the establishment of plantations to ensure that water is available for agriculture. A likely impact of lower rainfall in the east could therefore be further restrictions on forest establishment.

A wetter, more cyclone-prone west will not necessarily result in erosion-prone marginal farmland making way for planted forests. Repeated severe flooding and erosion events in the southern North Island have not led to a successful revegetation strategy. Instead, locals blamed damage to fences and bridges on forest debris washed down from harvested plantations. It would be reasonable to assume there will be some opposition to widespread afforestation of farmland, just as there has been in the past.

6.6 Summary: Best Estimates

It is important to remember that a planted forest is a reservoir of carbon but not necessary an active sink. There will be positive net uptake of atmospheric carbon as long as uptake due to growth is greater than emissions from decay and harvesting. In a normal forest with an equal area in each age class, net removals will be positive while the forest is expanding in area, but once harvesting begins, the forest will be neither a sink nor a source—the carbon stock will be maintained at a constant level. With any other age class distribution, the level of net removals will fluctuate, and if the level of harvesting is allowed to fluctuate, there will be periods with net emissions. The pre-1990 and 1990-2007 forests can be regarded as two separate forests with uneven age class distributions. Since 1985, the pre-1990 forest has been maturing—the average age and hence growing stock and carbon have been increasing. However, it is expected this forest will soon enter a phase where it is a net source for a decade. After this, it will again be a net sink as young, replanted stands mature and harvesting switches to the post-1989 resource. The cycle then repeats (Figure 12).

Under the scenario modelled, the post-1990 forest follows the same pattern, but with periods of net removals coinciding with periods when the pre-1990 resource is a net source. As a result, the combined resource declines from a large net sink to a small source.

7. Conclusions and recommendations

7.1 Present Status of Studies, Datasets and Analyses

In the past New Zealand has been able to meet its reporting obligations under the UNFCCC through the use of existing data sets and models. While the models have proven adaptable to a wide range of scenarios, there are a number of weaknesses, including:

- declining quality of NEFD area statistics
- accuracy and applicability of NEFD yield tables is largely unknown
- diversification of forest ownership and hence doubts over forest management, data reliability and growth assumptions
- inability to provide statistically based confidence limits with estimates

- lack of resolution for some analyses and lack of integration with other land use analyses.

LUCAS is designed to address these short-comings, but the move to an entirely different basis for inventory reporting is not without issues. At this stage we have no basis for assessing how well the current NEFD areas and yield tables reflect reality. LUCAS mapping and plot data will give us this information—initially for the post-1989 forests. In the meantime, the existing analyses serve as useful guides to expected trends in net removals and to the range of factors that can influence those trends. The models can be calibrated as LUCAS plot data (LIDAR and ground-based) becomes available—this would allow the current approach be followed for pre-1990 forest until a national set of inventory sample plots are established. Without the LUCAS data, there is a high level of uncertainty as to the actual magnitude of carbon stocks and stock changes we can expect.

7.2 Key Information Gaps, Uncertainties, and Research Priorities

The effect of information gaps and uncertainties on the above analyses has been largely taken into account through the range of scenarios presented. Further specific information gaps, for which likely scenarios cannot presently be proposed, are listed below.

The immediate requirement is for LUCAS planted forest data to be analysed and used to calibrate or replace both the post-1989 and pre-1990 NEFD-based yield tables as soon as possible. Verification of NEFD areas from LUCAS mapping work is also a priority. There are tasks required in support of the LUCAS approach as well as improvements to the NEFD-based approach that are required in the short-term:

- Verification of NEFD areas from LUCAS mapping, including the extent of over-planting in post-1989 forests
- Development of estimation functions from LIDAR metrics
- Calibration/replacement of post-1989 NEFD-based stand volumes with LUCAS plot data, including LIDAR
- Calibration/replacement of pre-1990 NEFD-based stand volumes with LUCAS plot data, including LIDAR
- Calibration of post-1989 NEFD-based dead-wood and litter pools with measured LUCAS plot data
- Analysis of stock changes following deforestation, including dead organic matter transfers to the soil carbon pool. Better characterisation of deforestation by area, age, pre-deforestation carbon stock, and treatment of residues, with appropriate decay rates
- Quantify the benefits of applying the A/R Debit rule to deforestation of post-1990 forests
- Understorey carbon stock changes following afforestation
- Douglas fir density and carbon partitioning (and to a lesser degree, eucalypts)
- Validation of harvesting assumptions regarding carbon removed offsite and residues remaining

- Radiata pine growth rate and density surface, to be applied to wall-to-wall mapping
- Mapping of forest management activities in planted forests using remote sensing techniques
- Improvements to historic time series (including 1990 baseline) based on better information on afforestation and deforestation
- Improvement of estimates of emissions due to pre-afforestation land clearance, using estimates of area planted by previous land use, and land use carbon stocks from the indigenous CMS
- Improvement of future harvest area estimates.

7.3 Implications of Accounting and Mitigation Options for New Zealand's Net Position

The growth of post-1989 forests makes a substantial contribution to New Zealand's net position in CP1. Including pre-1990 forests in the CP1 accounts, as would have occurred under an all-forests net accounting regime (but without a 1990 baseline), would reduce this by about 15%, or 9 Mt CO₂. The reduction would reach about 90%, or 70 Mt CO₂ in CP2, under an all-forests net accounting approach (again without a baseline), in comparison with continuing the current CP1 gross-net accounting regime. However, in CPs beyond 2023, including the pre-1990 estate will provide net removals, to help balance the net emissions from the post-1990 estate during periods in which harvesting is concentrated in the latter—and vice versa. The planted forest resource as a whole is nonetheless likely to be a small source of carbon under a net accounting approach (without a baseline) in future commitment periods unless there is:

- on-going afforestation or
- an increase in average carbon per hectare, through forest management (especially lengthened rotation ages).

The adoption of 1990 as a baseline for a net-net all-forests accounting approach provides a very high hurdle for New Zealand, and this is improved only slightly by using either a 5-year or decadal average. The models suggest that pre-1990 forests are unlikely to reach the level of removals achieved in 1990 again unless the rotation age is substantially lengthened. This is because it is assumed that harvest regulation will result in a more evenly spread age class distribution. Taken to the extreme, an age class distribution with equal areas in each age class up to the rotation age (i.e. a 'normal forest') will maintain a constant carbon stock with no net uptake or emissions.

The use of a "forward-looking" baseline would allow only improvements over and above "business as usual" to be accounted. This would mean that the 'hurdle' would not be the level of removals in 1990, but rather, the expected level of future removals from this resource. If this was not applied to both pre-1990 and post-1989 forests, there will not be sufficient qualifying removals to cover periods when the post-1989 forest is a net source. Neither would it provide a meaningful level of emissions offsets from New Zealand forests if applied to both pre-1990 and post-1989 forests, if the baseline under a forward-looking baseline approach was the level of removals achieved in the immediately prior commitment period. Overall, more details of proposals for a forward-looking baseline need to become available before its implications can be quantitatively assessed.

Table 13 Implications of accounting options on post-2012 accountable net carbon removals. Five-year commitment periods are assumed. The gross-net value is net of deforestation. Net All-forests adds in removals by pre-1990 forest. Net-net All-forests subtracts removals by pre-1990 forests at 1990 (without

attempting to otherwise “factor out” stand-age effects). More details need to be made available about the forward-looking baseline option before a quantitative assessment of that option can be provided

CP	Net Removals Per Commitment Period by Accounting Approach (Mt CO ₂)		
	Gross-net	Net, All-forests	Net-net All-forests (1990 Baseline)
1	62	54	35
2	77	9	-10
3	-3.3	-8	-27

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Appendix

New Zealand LULUCF Sector Overview

The LULUCF sector in New Zealand's greenhouse gas inventory is dominated by CO₂ removals by planted forests, which are estimated using a modelling approach and New Zealand-specific data. For all other land uses, the simplest IPCC methodology is followed, based on a land use change matrix. Removals and emissions from indigenous forest have been assumed to be equal in value. Five IPCC categories in the LULUCF sector are identified as trend and/or level key categories in the 2006 inventory (MFE 2008a, Table A1). Note that due to the limitations of the analysis, these results should be regarded as indicative until the 'wall-to-wall' land-use change mapping project is complete.

Table A1 LULUCF key categories in the 2006 New Zealand Greenhouse Gas Inventory

LULUCF category	Description	Contribution to Level (%)	Contribution to Trend (%)
Forest remaining land	Growth of planted forests from year of planting, net of harvesting	23.9	5
Land converted to forestland	Emissions from clearing land for planting (including decay of residues on previously cleared and planted land, but excluding planted forest growth)	2	9.4
Cropland remaining cropland	Growth of perennial woody crops	0.6	(Not a key category)
Liming emissions	Emissions resulting from application of lime to farmland and forestry	0.6	1
Land converted to grassland	Emissions from converting forest land to grassland	0.6	1.1

CHAPTER FOUR INDIGENOUS FOREST SINKS AND MITIGATION OPTIONS

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Summary

This Chapter deals with the contribution New Zealand's natural forests make to carbon stocks and stock changes, reported and accounted as part of the LULUCF sector under the United Nations Framework Convention on Climate Change (UNFCCC).

Changes in carbon stocks of indigenous forest in New Zealand are not presently included in the accounting of emissions or removals under the Kyoto Protocol unless these forests are involved in a land-use change. The inclusion of carbon-stock changes resulting from forest management was voluntary for CP1, and New Zealand chose not to include them. However, they are required to be reported under the UNFCCC. It is currently assumed these are old-growth forests that are carbon-neutral, although quantitative evidence for this is presently very limited.

In future commitment periods, emissions or removals by these forests may have to be accounted under some of the variants of all-forests or net-net accounting approaches. Because the area of indigenous forests is relatively large (c. 6 Mha), carbon-stock changes of even a few tonnes per hectare per year could have significant implications for New Zealand's carbon balance. Deforestation of indigenous forests is, however, minimal, and will result in little liability in future commitment periods.

The scope of the present study was to:

- provide a more robust assessment of likely changes in indigenous forest carbon stocks from analysis of inventory sample-plot data
- update decay rates of dead-wood in indigenous forests, as changes in the dead-wood carbon pool often contributes significantly for some decades to total changes in indigenous forest carbon stocks
- evaluate changes in carbon stocks with increased establishment of indigenous forest for emissions mitigation and erosion control on marginal lands
- document the key strengths, limitations, critical assumptions, uncertainties and knowledge gaps that are involved in making estimates and forecasts of indigenous forest carbon stocks
- identify and prioritise the knowledge gaps and uncertainties that most limit the reliability of the LULUCF component of forecasts of New Zealand's net emissions position.

Results from work completed in this component of this study are as follows:

- Carbon stocks of the indigenous forest live and dead carbon pools showed no significant changes over time. The study of change in the live biomass pool was based on a substantially larger set of inventory sample plots than in previous work (206 plots instead of 39). The results for the dead wood pool are the first reported, and are based on time-sequence studies on a set of 31 inventory sample plots that have at least 4 sets of measurements.
- Studies of the time-dependent change in the coarse woody debris (CWD) dead wood pool show that the average time taken for biomass to reduce by 50% (the decay half-life) is 30 years. The value was determined from studies using 7 indigenous tree species, of which 5 were among the 10 most abundant species by volume nationally in New Zealand. Decay rates

differed significantly between some of these 7 species; however, a mean decay rate is nonetheless appropriate, given that the species identity of CWD is often difficult to determine, and is often recorded in plot inventory data as unknown.

- Indigenous forestation of marginal lands offers considerable potential for both emissions mitigation, with co-benefits of erosion control, a more sustainable land-use, and increased indigenous biodiversity. Depending on the potential erosion severity rating that is used to define lands as “marginal”¹⁸, either 4.6 Mha or 2.7 Mha of marginal pasture lands are available—with about 60%, or 40%, respectively, of these lands in private ownership. Using indigenous forests only, forestation of all marginal lands would result in carbon sequestration over the active growth phase (at least 150 years) of 24 Mt CO₂ yr⁻¹ or 14 Mt CO₂ yr⁻¹, respectively, for the two classes of marginal lands. To place this in perspective, New Zealand’s present obligations under the Kyoto Protocol require additional offsets of 7.3 Mt CO₂ yr⁻¹ over the first commitment period.
- There are no major limitations, critical assumptions, large uncertainties or substantial knowledge gaps involved in making estimates and forecasts of existing indigenous forest carbon stocks. Although future cycles of LUCAS inventories will provide valuable confirmatory information, the analysis of data available to date has shown no significant change in live biomass stocks over time. Moreover, although our knowledge of carbon stocks in the dead-wood pools remains preliminary, it is unlikely these are changing significantly if live biomass stocks are also not changing. This is confirmed for such quantitative analysis as is possible to date (on just 31 inventory plots), although clearly a larger study is yet required before this can be fully confirmed. Any expanded study on dead-wood may have to be model-based however, as there are relatively few sites with a good temporal record of measured dead-wood stocks.
- The knowledge gaps and uncertainties that most limit the reliability of the estimate of carbon sequestration in re-established indigenous forests to NZ’s post-2012 net emissions position, and mitigation options, are:
 - The need for a more precise definition of marginal pasture land, and of the carbon price at which “carbon farming” on such land becomes a viable economic proposition—by region, and probably by land classes within regions.
 - Models of regional, and preferably sub-regional, rates of carbon sequestration in indigenous forests based on likely successional pathways.
 - Little current effort in developing and validating models of indigenous forest establishment, including of rates of canopy closure under natural regeneration regimes—and for establishment and growth of indigenous shrublands.
 - Lack of information on land management practices that can enhance natural regeneration rates for indigenous forest, and that can encourage rapid succession from lower biomass shrubland to high-biomass tall forest.

¹⁸ For this analysis, marginal lands were defined as grasslands in the LCDB2 dataset, less areas rated as woody in the Ecosat basic landcover dataset, that were below the treeline. All class 7 and 8 lands were considered marginal for pastoral farming, as well as class 6 lands with either a potential erosion severity rating in the NZLRI dataset of moderate to extreme, or severe to extreme.

1. Introduction

1.1 Scope of Study

Changes in carbon stocks of indigenous forest in New Zealand do not presently feature in accounting of emissions or removals under the Kyoto Protocol. However, they are required to be reported under the UNFCCC. It is currently assumed these are old-growth forests that are carbon-neutral, although this is supported by very little quantitative evidence. The possible implications of browsing pests on current carbon stocks and future forest regeneration is poorly understood.

The scope of the present study was to provide a more robust assessment of likely changes in indigenous forest carbon stocks, based on a national subset of sample plots originally measured under the older National Vegetation Survey (NVS) and remeasured as part of MfE's LUCAS programme. The study also included work on updating decay rates of dead-wood in indigenous forests, as this carbon pool has a much longer life-time (decades) than for exotic forests, and any decreases in live biomass, if there are any, could be significantly offset by gains in the dead-wood pool over the next several commitment periods. For completeness, the report also summarises and quantifies some emissions mitigation options using indigenous forests that are currently being considered by the Department of Conservation (DOC). This quantification includes an updated analysis on the areas of marginal lands that are potentially available for indigenous reforestation (i.e. with suitable seed sources present to promote natural regeneration).

1.2 Reporting and Accounting Requirements

In future commitment periods, emissions or removals by these forests would have to be accounted under any of the variants of all-forests net-net accounting approaches. Because the area of indigenous forests is relatively large (c. 6 Mha), changes in carbon stocks of even a few tonnes per hectare per year could have significant implications for New Zealand's carbon balance. Deforestation of indigenous forests is, however, minimal at present, and will result in little liability in future commitment periods.

Article 3.3 of the Kyoto Protocol allows the net changes in greenhouse gas emissions by sources and removals by sinks resulting from afforestation, reforestation and deforestation (ARD) since 1 January 1990 to be used to meet New Zealand's commitments. Emissions and removals must be measured as verifiable changes in carbon stocks in each commitment period. The carbon stocks to be accounted are above-ground biomass, below-ground biomass, dead wood, litter, and soil organic carbon. Emissions and removals from these same pools must also be reported under the UNFCCC in New Zealand's annual greenhouse gas inventory. However, the UNFCCC inventory reports carbon stocks in all forests (i.e. including pre-1990 forests) as part of the Land Use, Land Use Change and Forestry (LULUCF) sector.

2. Review of current data and models

2.1 Activity Data: Area of Indigenous Forests and Shrublands

The area of indigenous forest and shrublands in New Zealand was last updated at a relatively high level of thematic detail—42 indigenous forest, shrubland or mixed forest/shrubland/grassland classes—in 1987, with the publication of the Vegetation Cover Map (VCM) of New Zealand (Newsome 1987). Statistics from the VCM are given in Table 1 and, allowing for approximate

percentage cover in mixed classes, indicate a total area of indigenous forest of about 5.9 Mha, and about 2.4 Mha of indigenous shrubland. This compares relatively well with estimates from the New Zealand Land Cover Database (LCDB) of 6.3 Mha of indigenous forest, and about 2.7 Mha of shrubland (MAF 2002), although experience suggests the LCDB somewhat over-estimates the area of shrubland in montane areas.

The Crown is the major indigenous forest owner, with the Department of Conservation managing some 77% of the total indigenous estate for conservation, heritage and recreational purposes (MAF 2002). Twenty-one percent of the indigenous forest estate is in private hands, with the 2% balance in miscellaneous reserves (MAF 2002). The indigenous forest provisions (part IIIA) of the Forests Act 1949, introduced in 1993, require the sustainable management of privately owned indigenous forests. This means the forests are managed in a way that maintains their ability to provide products and amenities in perpetuity.

Table 1 National indigenous forest and shrubland areas, from Tate et al. (1997).

Vegetation Class	Area (Mha)
Shrubland	1.092
Mixed shrubland/grassland	5.090
Mixed forest/shrubland	1.285
Forest	5.120
Mixed forest/grassland	0.731

2.2 Carbon Stocks in Forests and Shrublands

Carbon stocks in New Zealand's indigenous forests and shrublands were first estimated by Tate et al. (1997), and biomass carbon stocks in the major forest classes have recently been estimated independently for the DOC estate (Carswell et al. 2008) based on analyses of data in the National Vegetation Survey (NVS—see next section below) database. Table 2 gives values for the national carbon stocks in indigenous forest and shrubland biomass (i.e. in the vegetation, dead-wood, and litter pool), taken from Tate et al. (1997). Values for the soil carbon pool are taken from Scott et al. (2002), as given by Carswell et al. (2008). As expected, carbon stocks in the soil can considerably exceed those in vegetation biomass for areas with limited woody cover. Table 3 gives the most recent values for indigenous forest biomass in the DOC estate, taken from Carswell et al. 2008. Table 3 gives the values for biomass carbon stocks in the DOC estate, for the major forest types.

Table 2 Estimates of national indigenous forest and shrubland carbon stocks. The biomass pool values were summarised from data provided by Tate et al. (1997), and soil carbon pool values from Carswell et al. (2008). The biomass pools considered are above- and below-ground live biomass, coarse woody debris and forest-floor litter.

Vegetation Class	Carbon Stocks (Mt C)	
	Biomass Pools	Soil Pool
Shrubland	76	173
Mixed shrubland/grassland	110	750
Mixed forest/shrubland	230	201
Forest	1710	797
Mixed forest/grassland	72	116

Table 3 Estimates of indigenous forest carbon stocks in the DOC estate, from data provided by Carswell et al. (2008). The biomass pools considered are above- and below-ground live biomass, coarse woody debris and forest-floor litter

Vegetation Class	Carbon Stocks in the Biomass Pools (Mt C)
Podocarp forest	136
Lowland podocarp–broadleaved forest	222
Highland podocarp–broadleaved forest	159
Lowland podocarp–broadleaved–beech forest	253
Highland podocarp–broadleaved–beech forest	171
Beech forest	204
Beech–broadleaved forest	237
Broadleaved forest	196

2.3 The NVS Database: Summary

The National Vegetation Survey Databank (NVS—“Nivs”) is a physical archive and computer database containing records from approximately 45 000 vegetation survey plots, including data from over 12 000 permanent plots (<http://nvs.landcareresearch.co.nz/>). NVS provides a unique record, spanning more than 50 years, of the characteristics of indigenous and exotic plants in New Zealand's terrestrial ecosystems, from Northland to Stewart Island and to the Kermadec and Chatham Islands. A broad range of habitats are covered, with special emphasis on indigenous forests and grasslands. Data in the NVS is comprised of surveys conducted by the New Zealand Forest Service, Department of Lands & Survey, and the DSIR Botany Division, and ongoing surveys and research by the Department of Conservation, regional councils, universities and Landcare Research. Such widely sourced information collated in one database is part of the value of NVS to New Zealand. Landcare Research attempts to hold the most up-to-date and complete copies of data in NVS, but does not guarantee that all data are error-free. Data are frequently updated or corrected by users during data analyses or plot re-measurement. Data collection, storage and quality control procedures are summarized elsewhere by Wiser et al. (2001), Payton et al. (2004) and Hurst and Allen (2007a, b).

NVS contains a range of vegetation data, including:

- general vegetation survey data (reconnaissance—RECCE—descriptions) from plots that are usually not permanently marked (Hurst & Allen 2007b)
- permanent plot data where fixed area plots or transects have been established, and the vegetation has been measured precisely (e.g., tagged trees, sapling and seedling counts, species lists; Hurst & Allen 2007a). Fixed area plots may be circular, square quadrant, rectangular, or cruciform. Inventories at most of these plots follow standard methods, e.g., in forests all trees (i.e. woody stems ≥ 2.5 cm diameter at 1.35 m (DBH)) within a fixed quadrant area (usually 400 m², 0.04 ha) are permanently tagged to allow repeat measurements. Most plots are located along objectively located transects.

Data within NVS support reporting requirements for the Convention on Biological Diversity, United Nations Framework Convention on Climate Change, Resource Management Act, State of the Environment, and the Montreal Process. The data also assist planning of resource management and ecological restoration. Historical information in the NVS is significant in enabling New Zealand to address issues of current concern that were unforeseen at the time of data collection. They include assessing the impacts of climate change on indigenous ecosystems, setting restoration goals in areas that have since degraded, and the storage of carbon in indigenous ecosystems. For example, NVS plots have been essential for assessing carbon storage in New Zealand's indigenous forests (Hall & Hollinger 1997), and for designing a system for monitoring carbon in New Zealand's indigenous forests and shrublands (Coomes & Beets 1999, Coomes et al. 2002)—which ultimately became the Carbon Monitoring System implemented by the Ministry for the Environment (now part of LUCAS).

2.4 The Indigenous Forest LUCAS/CMS Dataset: a Summary

The MfE-funded Carbon Monitoring System (CMS), now called LUCAS, was a 5-year data collection programme that commenced in 2002. It established a national grid (8×8 km) of permanent plots to make repeated measurements of indigenous forest and shrubland carbon stocks (Coomes et al. 2002; Payton et al. 2004). CMS plots were located on an 8-km grid, where the intersections of grid lines occurred in vegetation defined either as shrubland or indigenous forest by the Land Cover Database (LCDB1). At each location, a 20×20 m plot was laid out, with the data for estimating carbon stocks in woody vegetation recorded following the methods outlined in the CMS field manual (Payton et al. 2004). Data on species, and DBHs and heights, have been collected from 1260 forest and shrubland plots, and are known to represent a systematic, unbiased sample. Plot sheets are stored in the National Vegetation Survey (NVS) databank maintained by Landcare Research at Lincoln. Some of these data have been entered and collated by Landcare Research into an electronic database, but the majority of data were entered, and are held by, Interpine Ltd (<http://interpine.co.nz>) on behalf of MfE.

There is on-going error checking of the CMS data. MfE grants access to the CMS data “with no assurance being given to the quality or completeness” of the data. Although Landcare Research has recently completed error checking of the RECCE data (i.e. vegetation composition information), error checking of other data, including tree diameters, is not complete. For tree diameter data, most data quality issues raised by Interpine related to duplicate tree tags for individual stems (48 908 cases). Preliminary error checking by Landcare Research, suggests that error rates in tree stem status are about 0.3% (A. Marburg, pers. comm.), and for DBH may be up to 3% (Andrea Brandon, MfE, pers. comm.). Despite the need for on-going error checking, the CMS dataset is currently the best unbiased and most representative dataset for assessing live biomass carbon stocks in indigenous forests and shrublands at a national scale.

2.5 Carbon Stocks Estimation Equations for Above-ground Biomass: Summary

Previous calculations of live biomass C stocks in trees have used diameter to estimate tree volume using allometric relationships between diameter, height and volume. This approach is summarised by Coomes et al. (2002) and Peltzer and Payton (2006). Because NVS plots did not contain tree height data, tree height needed to be derived from the measured diameter at breast height of 1.35 m, DBH, using the relationship of Peltzer and Payton (2006), which meant that the diameter was “double sampled”.

2.5.1 Height measurements and modelling tree height from diameter

On each plot where they were present, the height of the top of the crown (and DBH) of 15 individuals from each of the following groups: broadleaved trees, conifers, tree ferns and dead standing stems were measured. The sample included the full diameter range for each group, and included malformed stems. The sample also included all trees > 60 cm DBH or, where trees of this size were absent, the largest five trees on the plot. Where stems leaned more than 20° from vertical, lean angle was measured to the nearest 10°. Height was corrected by dividing the measured height by the cosine of the lean angle (where this angle was expressed in radians).

These height measurements were used to model the relationship between diameter at breast height and tree height. Two types of relationship were explored between diameter and height – linear (1) and log-linear (2):

$$\text{Height} = a + b \bullet \text{DBH} \quad (1)$$

$$\text{Height} = a + b \bullet \ln(\text{DBH}). \quad (2)$$

The relative ability of either model to predict height was assessed using the corrected Aikake information criterion (AICc):

$$AICc = N \ln\left(\frac{RSS}{N}\right) + 2K\left(\frac{N}{N - K - 1}\right) \quad (3)$$

Where: N is the number of cases or replicates, RSS is the error sum-of-squares and K is the number of parameters included in the model.

An Akaike weight, which gives an estimate of the probability that a model gives the most parsimonious fit to the data, was calculated for each model following the method described by Johnson and Omland (2004):

$$W_i = \frac{\exp\left(-\frac{1}{2} \Delta_i\right)}{\sum_{j=1}^R \exp\left(-\frac{1}{2} \Delta_j\right)} \quad (4)$$

Where: R is the number of models under consideration and Δ_i is the difference between the $AICc$ value of model i and the minimum $AICc$ value across all models. The sum of W_i values across all models adds to unity. The log-linear model was considered to be the default diameter-height relationship, while a linear model was selected if it had $W_i > 0.7$.

This process was performed for each species with > 30 height measurements. For all other species, height was estimated following the relationship between diameter and height for all stems where height measurements were taken. Where a species had > 30 stems and there was no evidence for a relationship between diameter and height ($p > 0.15$ for both linear and log-linear models), each stem was assigned the species' mean height. Modelled heights were applied to all stems where no measured height was recorded, while measured stems retained their measured height value. The coefficients derived from these models were used to:

- (i) estimate heights for individual trees from DBH measurements;
- (ii) estimate the volume of tree components (i.e. stem, branches and foliage), based on the DBH/height data and the allometrics of Beets et al. (2001) and Coomes et al. (2002);
- (iii) convert the volume to biomass by applying basic wood density (kg/m^3) collated by Ian Payton (unpublished data) and Peltzer and Payton (2006);

- (iv) convert biomass to carbon using a carbon fraction of 0.5.

Only tree stems that were classified as either “alive” or “dead” were included in the analyses of C. Stems “not found” (i.e. trees previously tagged in NVS plots but not in the CMS plots) and “unknown” stems were excluded, but comprised less than 2% of total tree stems. No epicormic or epiphytic stems were included in these analyses. No adjustments for biomass or C content were made for fused or multiple-stemmed individuals.

2.5.2 Estimating carbon in live trees

Trunk volume was estimated as a product of DBH and height following the allometric relationship of Beets (1980):

$$\text{Volume} = 0.0000598 (\text{DBH}^2 \bullet \text{Height})^{0.946} \quad (5)$$

The biomass contained within each trunk was calculated as:

$$\text{Stem mass} = \text{Wood Density} (1.0 - 0.0019 \bullet \text{DBH}) \times V \quad (6)$$

Wood density values were obtained either by Ian Payton (as cores or discs) or from Peltzer and Payton (2006). Where values were available from both sources, the former was chosen as these derive from a consistent methodology. Where multiple values for a species were available from this source, the mean was taken. For species where no density value was available, the mean density taken across all species was assigned. Most dominant tree, shrub and tree fern species, including all *Nothofagus* (beech) species, have published values of wood density. These species collectively comprise more than 90% of the total biomass in forest plots.

The biomass of branches >10cm diameter and foliage was estimated as:

$$\text{Branch mass (kg)} = 0.03 \times \text{d.b.h.}^{2.33} \quad (7)$$

$$\text{Foliage mass (kg)} = 0.0406 \times \text{d.b.h.}^{1.53} \quad (8)$$

Root mass was assumed to be 25% of the live aboveground tree biomass. Finally, tree biomass was assumed to be comprised of 50% carbon (Coomes et al. 2002).

The use of allometric relationships to obtain volume, biomass and carbon estimates has generated considerable debate in the forest inventory and carbon calculation literature that remains unresolved (see Chave et al. 2004). Here we apply universal volume functions for predicting aboveground components of live trees (i.e. stem, branch matter > 10 cm in diameter, twigs and leaves) as described by Hall et al. (1998) and Coomes and Beets (1999), based on data in Beets (1980). These universal allometric functions are appropriate for large-scale assessments of C (Beets et al. 2001). For example, errors associated with misapplying adult tree functions to juvenile trees are largely due to ontogenetic changes in wood:foliage ratios, but this error is small because foliage comprises less than 5% of the tree C stocks for stands on average. Similarly, coefficients of a volume function based on DBH and height can vary from 10 to 15% among plant species, but can nevertheless be used for national scale assessments because forests are comprised of mixed species, with compensation expected to occur (Beets et al. 2001).

An audit of 178 randomly selected CMS plots revealed that 185 stems > 2.5 DBH recorded on plot sheets had not been entered into the CMS database. This equates to 1.13% of the stems entered for these plots. On average, this resulted in 1.15 t C ha⁻¹ of carbon missing from the database per plot, causing a total downward bias of 0.78% in the estimation of live carbon stocks. There was no relationship between recorded carbon stocks and the amount of carbon missing across the audited plots. Therefore, this bias was corrected by adding 1.15 t ha⁻¹ of carbon to the live stems > 2.5 cm DBH pool, and to total carbon content for all non-audited CMS plots. Audited CMS plots received the amount of carbon estimated to have been omitted from the database during the audit.

Above-ground biomass and C contents of live trees were converted from kg/0.04ha to t/ha by multiplying by 0.025 (i.e. converting mass or C expressed as kg/plot area to t/ha), and then cosine correcting these totals for plot slope (i.e. by dividing measured carbon by the cosine of the slope angle). Here we report results for live tree biomass C pools only, and do not provide estimates of other pools including coarse woody debris, shrubs, soils or litter.

3. Estimates of change in live biomass stocks

3.1 Reprocessing of the CMS and NVS Data

Of the 986 CMS plots that were located in indigenous forest (i.e. recorded as such by the CMS field crews), 206 were previously established as NVS plots. Original measurements on the NVS plots were typically made 20-25 years ago (mean c. 20 years). These remeasured plots allow for calculation of change in time of live tree biomass C stocks, and are the basis of the analyses here. However, comparisons of live biomass C in remeasured CMS plots with the original NVS data are not possible for two main reasons:

- (i) Tree tag matching had not been done between NVS and CMS plots.
- (ii) No error checking of the CMS diameter data, and many of the original NVS plots, had been undertaken.

Clearly, tree tag matching and data error checking are necessary before any meaningful calculation of change in tree demographic contribution to biomass or C change between NVS and CMS re-measurements could be undertaken. We therefore reprocessed all data on trees from remeasured NVS/CMS plots located in indigenous forest. Trees were defined as tagged stems (i.e. stems having a DBH of ≥ 2.5 cm).

The approach taken was to conduct a line-by-line check of the entered data for *all* NVS and CMS measurements against the original field plot sheets. This is a relatively large task. Tree tag numbers are matched between all previous NVS measurements and the CMS measurement and updated in the NVS system as the most recent tree tag number. At the same time, error checking was carried out line-by-line to ensure that the correct data as recorded on the field sheet had been entered. Errors that were encountered included: diameters measured at other than DBH or tag height; missing tree tags; tags on remeasured plots (presumably replaced tags) not always being at standard breast height (though permanent plot protocols allow for this if the diameter is “irregular” at breast height; Hurst & Allen 2007b); incorrect species identification; inconsistent synonyms used for species identity; incorrect diameter values entered; and incorrect stem status (i.e. alive, dead or missing). Stem status is critical for information needed to calculate demographic rates (i.e. tree turnover; stem recruitment and mortality) and partitioning changes into C pools resulting from tree establishment, growth and mortality.

Database software was developed by Landcare Research to facilitate the data reprocessing task by enabling side-by-side checks of NVS and CMS data to be made electronically, with logging of all changes or corrections made to the data. This captured log of changes is stored separately from the NVS and CMS data, rather than replacing the original data, and is then applied as a patch to a separate version of the NVS or CMS data. Such error checks will eventually need to be incorporated in a date-stamped version of the NVS and CMS datasets.

3.2 Estimates of National Stock Change

3.2.1 Results of previous work

For completeness, and to provide a basis for comparison of the results from different sample sizes, we provide here a summary of an earlier study (Peltzer & Payton 2006) comprising analysis of repeat measurements at the 39 NVS/LUCAS sample plots measured in the first year of inventory, beginning in 2002.

Live biomass C of trees was, on average, 24.1 t C/ha lower in the CMS measured in year one (i.e. the most thoroughly error-checked) than at the original NVS plot measurements. However, total-C was not significantly different between NVS and CMS measurements on these plots (t_{76} df=1.022, $P = 0.31$), and the statistical power of the test for differences in mean total-C between plots was low (0.052). A total of 290 plots would be needed to detect a significant difference between NVS and CMS measurements, given the relatively small difference in mean responses and high variability in total-C among plots. Alternatively, given 39 NVS plots and the variability among these plots, the least significant value (i.e. the minimum difference that is detectable given the sample size and variance in the data) is 46.9 t C/ha.

Nevertheless, the trend of declining total-C was due to C losses from tree mortality exceeding gains through tree growth and recruitment of new stems. There is no consistent trend of declining C across the NVS plots, but rather C losses were caused by disturbance removing the majority of live tree C at a small subset of plots. This is a predictable pattern because biomass gains through growth and recruitment are driven by relatively small annual changes whereas mortality that kills individual large trees or many trees at the plot scale reduces live tree biomass profoundly and immediately.

Table 4 Summary of mean tree mass and C in 206 remeasured NVS plots, and the mean annual C fluxes from growth, mortality and recruitment of trees

Parameter	Mean	SE	95%CL
NVS Total live tree-C (t C/ha)	209	16.6	32.6
Total-C change (t C/ha/yr) from persisting trees*	1.13	0.33	0.65
Total-C change (t C/ha/yr) from dead trees**	-2.86	0.54	1.06
Total-C change (t C/ha/yr) from new trees***	1.22	0.27	0.53

* Trees alive at both NVS and CMS sampling

** Trees alive for NVS measurement but recorded as dead in CMS measurements

*** Trees were recorded for the first time in CMS measurements

Thirty-nine of the year one CMS plots measured were previously established as part of the National Vegetation Survey (NVS) databank. Changes in live tree C between NVS and CMS measurements (typically 20 years) were partitioned into tree growth, mortality and recruitment. These analyses revealed that live stems on the remeasured plots had, on average, lost 0.51 ± 0.9 t C/ha/year with new tree growth of 2.35 t C/ha/year being offset by C losses through tree mortality of 2.86 ± 1.1 t C/ha/year (Table 4). These changes were not statistically different from zero. Closer inspection of C changes within individual plots revealed that no consistent trends of decline were occurring across NVS plots, but rather C losses were caused by disturbance removing most plant biomass from a few plots.

The comparison above does not include dead wood, soil C or litter, which would be expected to change little or even increase following disturbance. In fact, since it is major disturbance at a few sites that results in most of the decrease in carbon stocks, and given slow decay rates of most indigenous species, it can be reasonably assumed that there is little change in mean carbon stocks averaged over all biomass pools.

3.2.2 Results from this study

180 NVS plots were remeasured as indigenous forest plots during the first full measurement of CMS/LUCAS, and provided 20 534 tagged tree stems recorded as “alive” for these analyses. Mean (s.e.m.) live tree biomass C was 204.3 (10.2) t C /ha for original NVS measurements, and 225.6 (9.51) t C /ha for the CMS re-measurement, although this difference was not significant ($t_{1,358} = 1.53$, $P = 0.128$). A scatterplot of NVS and CMS biomass C estimates showed a strong, positive relationship between NVS and CMS measurements with few plots deviating from a 1:1 relationship (Fig. 2). Similarly, the frequency distributions of live biomass C for NVS and CMS measurements are very similar (Fig. 1). The statistical power to detect a difference in tree biomass C stocks between measurements was low ($\alpha = 0.05$, power = 0.3309). Given the sampling design and variation in biomass C standing stocks, the least significant value of detectable change in live tree C is 27.5 t C /ha. Given the variation and observed difference in biomass C between NVS and CMS re-measurements, the least significant number = 596.3 (i.e. the number of remeasured plots that would be required to produce a significant result).

Figure 1 Frequency distribution histograms of live tree biomass C for 180 permanent 20 × 20 m plots located in indigenous forest for original NVS surveys (top panel) and remeasured during CMS/LUCAS (bottom panel). NVS: mean = 204.3 (Confidence limits 184.1 - 224.5) t C/ha; CMS: mean = 225.6 (Confidence limits 206.8 t - 244.4) t C/ha

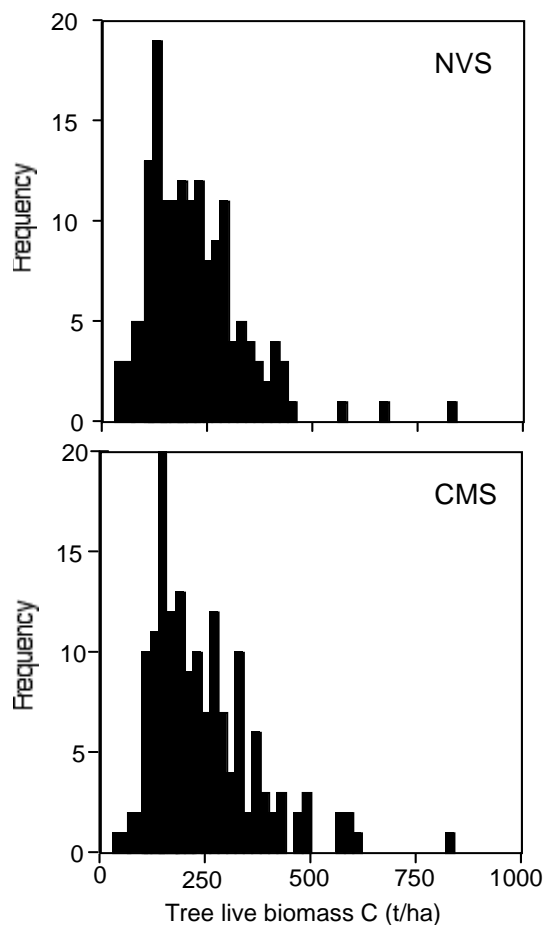
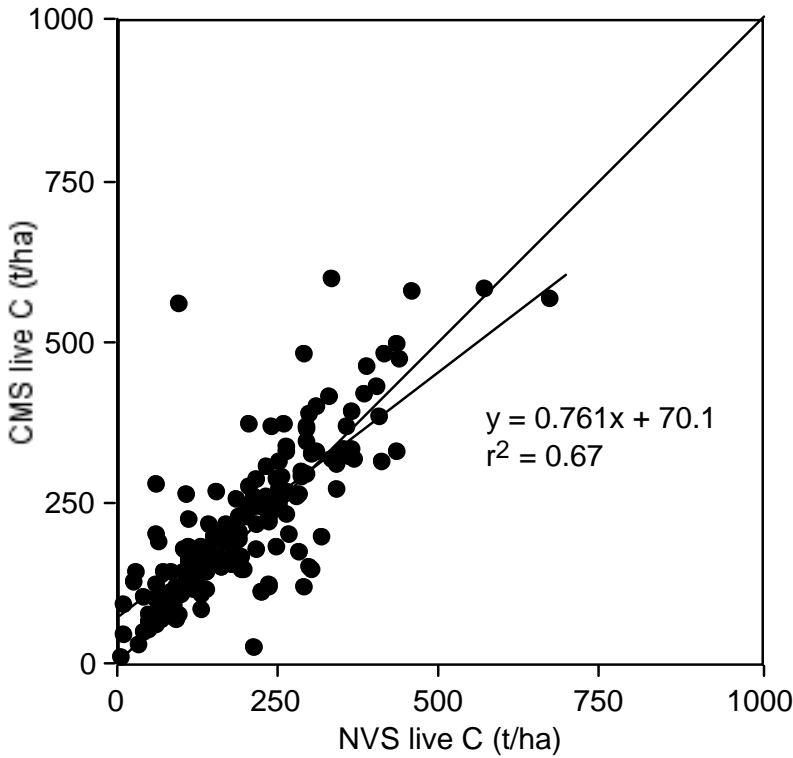
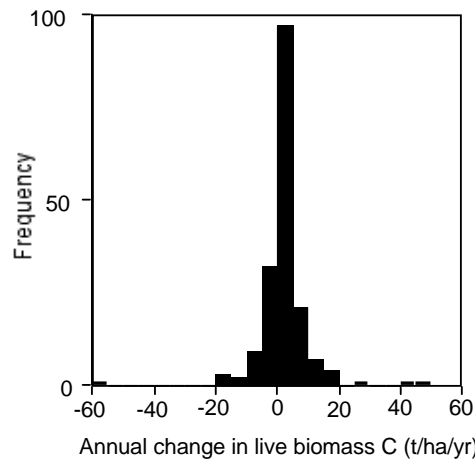


Figure 2 Scatterplot of CMS and NVS live tree biomass C. Linear regression shows a strong relationship of biomass C between initial (NVS) and re-measurement (CMS). A small number of plots departed markedly from the 1:1 line (diagonal line shown)



The mean time between NVS and CMS measurements was 17.4 yr (range from 1 to 32 yr). The mean (95% CL) annual change in live tree biomass C between measurements was 1.98 (1.24) t C/ha/yr. The frequency distribution of live biomass C change strongly peaks at 0–5 t C/ha/yr, with an interquartile range of -0.27–3.60 (Fig. 3). Similarly, numbers of tagged live tree increased by 2.42 (± 1.11) stems/plot/yr from NVS to CMS measurements. These results suggest that the modest (but not statistically significant) gains in C between NVS and CMS re-measurements are likely due to both recruitment of new tree stems into plots, and growth of existing stems between measurements.

Figure 3 Frequency distribution histogram of annual change in live tree biomass C for 180 permanent 20 × 20 m plots located in indigenous forest. Mean = 1.98 (Confidence limits 0.73–3.23) t C/ha/yr



At the first measurement, the diameter (D1) at 1.35 m (DBH) of all stems ≥ 2.5 cm DBH in each plot was recorded and each tree was uniquely identified with a numbered tag nailed to the stem at the height of the diameter measurement. When the plots were revisited during CMS, usually after an interval of more than 5 years, the diameters of all tagged stems were remeasured, any newly recruited stems (i.e., those at least 2.5 cm DBH) found in the plots were tagged, and any dead stems noted (see error checking details above). These data would allow an analysis of the change in aboveground C for live trees and changes in C caused by tree growth, mortality and recruitment over c. 20 yr since the NVS plots were first established.

We could not complete a demographic analysis of biomass and C change partitioned into tree recruitment, growth or mortality. This was for two reasons. First, there were errors in the diameter data that require further error checking. For example, about 200 stems were missing from the dataset, and a further c. 1600 stems have diameters that appear to be outliers, are undersized (i.e. < 2.5 cm dbh) or require error checking; this represents 8–9% of the data. Second, although a major effort has gone into matching tree tags between NVS and CMS plots, there remains additional work to complete the back corrections. For these reasons, it is premature to complete a demographic analysis of change in tree biomass C pools. We are currently completing these error checks and tree tag back corrections in related research projects, and anticipate these corrections to be completed in early 2009.

3.3 Uncertainty/Sensitivity Estimates

The following improvements could be made to reduce the uncertainty/error of estimates:

- (i) Improved allometric relationships relating tree diameter and height to biomass are needed, particularly for dominant tree species. The allometric relationships used by Coomes et al. (2002), and in this report, are based on data from Beets (1980) collected at a single site. Species variation is allowed for only through variation in wood density. Additional allometric relationships are needed particularly for the most common 20 species, which comprise the great majority (i.e. $> 90\%$) of the C in New Zealand's indigenous forests. For the purposes of C accounting, better allometric relationships are particularly important for predicting species-specific stem volumes and branch biomass (which typically comprise 56–67% and 12–19% of C in mature New Zealand forests respectively; Beets et al. 2001). The highest priority should be given to predicting the stem volumes of the largest trees (i.e. those > 60 cm DBH); the current generic allometric relationships did not include trees of this size. Although species-specific weighting factors could be applied in order to eliminate any bias expected to occur when forest composition or environment (e.g., elevation) varies, an approach independent of such factors is deemed appropriate for national-scale assessments of biomass or C (Beets et al. 2001).
- (ii) All estimates of C stocks in live trees rely on accurate estimates of basic wood density. Published values for some species can vary by up to 20% (c.f. the about 5% suggested in Beets et al. 2001; Table 5.4). This suggests wood density will likely vary substantially across indigenous species. This variation can be caused by differences within a single tree (i.e. from heartwood to sapwood), among trees within a site, or among sites. For example, wood basic density can vary from 100–150 kg/m³ between trees within one site (Beets et al. 2001). A much more comprehensive set of wood density measurements are needed for the major indigenous tree and shrub species, including sufficient data to determine the significance of density variations at the regional scale.

4. Initial estimates of change and dead wood stocks of tall forest

4.1 Introduction

The LUCAS natural forest plot network was installed on an 8-km grid over a 5-year period from 2002 to 2006, and has been measured once. Approximately 200 National Vegetation Survey (NVS) plots established 20–40 or more years before the start of the LUCAS project were incorporated and remeasured as part of LUCAS, to provide an indication of whether New Zealand's natural forests are carbon neutral. Coarse woody debris, fine litter, and soil carbon have only been measured once, when the LUCAS plots were installed and measured.

The subset of NVS plots, however, have been remeasured regularly and provide time series data for estimating the above ground carbon stock of individual trees at the time when tree mortality likely occurred, using allometric equations (Beets et al. 2008a). This information can be combined with the coarse woody debris (CWD) carbon stock estimate obtained at the last plot measurement date, to indirectly estimate the change in the coarse woody debris pool, using recently developed decay functions for indigenous trees in mature natural forest (Beets et al. 2008b). Only NVS plots with 4 or more measurements were selected for analysis.

4.2 Reprocessing the CMS Data

4.2.1 Data and analysis

When NVS plots were first installed and measured, all live trees that attained 2.5 cm DBH were tagged and DBH measured. Tree heights were usually not measured until these NVS plots were measured as LUCAS plots from 2002 to 2006. Tree tag numbers could not be used to track individual trees over time, because a variable number of characters of the tag number (typically the last 3–4 digits) had been entered electronically. Trees within the 20 × 20 m NVS plot were therefore manually checked and assigned a consistent identification number over time.

Before determining the approximate year when a tree died, various data issues needed to be resolved. For example, the tree number assigned to a tree needed to be consistent over time, and the tree status code (live or dead) needed to be checked and assigned if necessary. The amount of data checking involved was relatively large and time consuming, and it was therefore considered necessary to restrict the analysis to trees that attained 25 cm or more in diameter at breast height (DBH). The subset of trees that attained 25 cm in DBH at one or more of the measurement dates represented approximately 10% of the total number of trees measured in these plots. Nevertheless, these trees represent the source of the bulk of the CWD, which is defined by LUCAS as material > 10 cm in diameter.

In general, the approach taken resulted in satisfactory estimates of survival and mortality over time of individual trees in each plot. When a tree disappeared from subsequent records, it was assumed to have died midway between the last recorded measurement as a live tree and the subsequent measurement, and a record was inserted with a dead status code assigned. However, in some cases, for example when land movement resulted in trees being displaced out of the plot, it was assumed they had died, possibly contributing errors to the mortality estimates. There were also cases where trees were missed at some measurement dates and an interpolated DBH needed to be inserted before live stocks and changes could be calculated. In other cases, a large tree was measured and recorded as alive at the LUCAS measurement for the first time. Such trees may have existed as dead spars dating from the time the NVS plot was first installed and measured, which would explain why they were not (usually) measured by NVS teams. Alternatively, the LUCAS teams unlike the NVS team

may have decided that a particular tree was within the 20 × 20 m plot boundary. We assumed the latter, so that such trees were effectively excluded by us when calculating CWD. It was also necessary to correct some obvious outlier DBH measurements made evident by examining individual tree growth trends.

4.2.2 Uncertainty and sensitivity analysis

Standard errors and 95% confidence intervals were obtained for estimated mean CWD stocks in 1990 and at the LUCAS measurement date, and the stock change was calculated using the between-plot variation. These uncertainty estimates only account for plot-to-plot variation and exclude any errors resulting from the functions (e.g., volume functions, height/diameter functions) and parameters (e.g., wood densities, decay constants) involved in the calculations. The above calculations were also performed using 1980 rather than 1990 as the base year to test the sensitivity of the approach.

A number of models with empirically determined model coefficients were used in estimating the CWD carbon stock changes in this report. To test the sensitivity of stock change estimates to possible errors in these models, mean stock change estimates were obtained with several important parameters varied by ± 10% of their assumed true value. The parameters tested in this fashion were as follows:

- The decay constant. A common value of -0.0229 was used for all species in this analysis. The effect of varying this by ± 10% was examined.
- The CWD decay class modifiers. These indicate the reduction in density due to decay and are tabulated for only 4 species with all other species using common values. Values range between 1 (decay class 0, all species) and 0.33 (decay class 3, Red Beech). Reductions in density due to decay were varied by ± 10% of their mean values.
- Wood density. These are tabulated for a range of species. All wood densities were varied by ± 10% of their tabulated values.
- CWD wood density. Wood densities of CWD material were varied by ± 10% of their tabulated values. However, densities of trees that died during the defined period were not varied from their tabulated values.
- Stem and log volumes. All calculated volumes of mortality trees, spars and logs were varied by ± 10% of their calculated values.
- CWD volumes. All calculated volumes of spars and logs in the CWD but not of trees that died during the defined period were varied by ± 10% of their calculated values.

4.3 Estimation of Dead Wood Stocks, Change and Decay

Species/plot specific height/DBH functions were derived using the live trees sampled for height. A number of nonlinear height/diameter functional forms were tested, and the most suitable form chosen. This functional form enabled the height-diameter relationship to be transformed into a linear form. A model was then fitted to the transformed data with plot and species specific parameters enabling heights to be predicted from the DBH for live trees given the species and plot number. These height/diameter functions were subsequently used for estimating volumes of

complete trees in the year they were estimated to have died, and of standing dead spars as explained below.

A decay function was applied to indirectly estimate the CWD carbon as at the last plot measurement date of trees that died post-1989. The decay function was developed using 7 indigenous tree species, of which 5 were among the 10 most abundant species by volume nationally in New Zealand (Beets et al. 2008a, b). Decay rates differed significantly between some of these seven species; however, the mean decay rate across all seven species was considered appropriate for the current study, given that the species identity of CWD assessed at the last plot measurement date was often classified as unknown. The average decay rate for the 7 species, expressed as the time taken to lose 50% of the dead-wood biomass ($t_{0.5}$, the decay half-life) was 30 years. Larger trees (90 cm DBH, $t_{0.5} = 38$ years) decayed more slowly than small trees (DBH 30 cm ; $t_{0.5} = 21$ years). The slowest decay rates were recorded for *P. taxifolia* (matai; $t_{0.5} = 39$ years), and the quickest for *D. dacrydioides* (kahikatea; $t_{0.5} = 14$ years).

Coarse woody debris per plot included standing spars and fallen logs measured at the final LUCAS measurement date. Volumes of standing spars were estimated using a taper model developed for Douglas-fir which uses the DBH, the original total height of the tree, and the measured truncated height of the spar. Volumes of fallen logs were estimated using log diameter and length data of individual log sections using the truncated cone volume formula. Both types of CWD were converted to carbon using breast height outerwood density as tabulated for live trees by species (Beets et al., 2008a), and the decay class assigned by field team was applied to account for the density reduction due to the state of decay. Individual spar and log carbon estimates were summed per plot to provide an estimate of the CWD carbon stock at the time of the LUCAS measurement.

Above ground carbon of trees that died during the period from 1990 to the and the LUCAS measurement date was estimated from the DBH, and the predicted tree height, and species specific outerwood density using a generalized allometric equation given in Beets et al. (2008a). The loss in carbon due to decay from the estimated time of tree mortality based on the historic plot data to the LUCAS measurement date, was estimated using a decay function given in Beets et al. (2008b).

Carbon contributed to the CWD pool from 1990 to the LUCAS measurement date was subtracted from the LUCAS CWD stock estimate for each plot. This residual CWD estimate was assumed to have persisted since 1990. The estimated proportional loss in mass over the period from 1990 to the LUCAS measurement date was obtained using the decay function. To estimate the CWD stock as at 1990, the residual CWD was divided by this proportion.

The change in CWD carbon stock from 1990 to the LUCAS measurement date was calculated by subtracting the CWD carbon stock as estimated at 1990 from the LUCAS CWD carbon stock for each plot. The average change in CWD carbon stock per plot was scaled assuming plots were 20 × 20 m in area, to give a national CWD stock change per ha. Annual stock change estimates were obtained by dividing the total change by the number of years between 1990 and the LUCAS measurement date.

4.4 Initial Estimates of Dead Wood Stocks and Change

4.4.1 Stocks and change

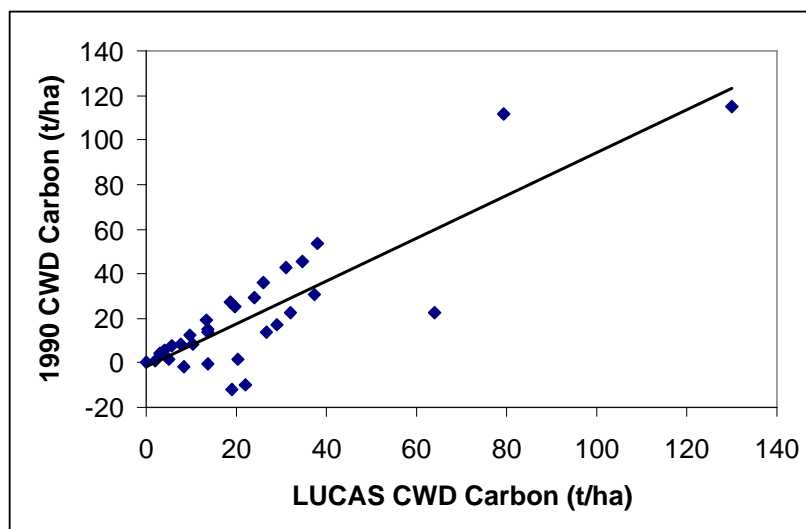
Total CWD carbon for the LUCAS measurement averaged 25 t/ha, with about two thirds stored in fallen debris (> 10 cm diameter) and one third in standing spars (> 25 cm DBH) (Appendix 1). On average, 2.1 trees (> 25 cm DBH) per plot were recorded as having died during the period between

1990 and the LUCAS measurement date (Appendix 2). The estimated contribution to the LUCAS CWD carbon stock from these trees averaged about 9 t/ha (Appendix 2), more than one third of the total of 25 t/ha. This was subtracted from the total LUCAS stock to estimate the carbon in 1990 using the decay model. The estimated CWD carbon stock in 1990 averaged 22 t/ha, slightly less than at the LUCAS measurement (Appendix 3).

In 4 of the 31 plots, the carbon from tree mortality over the period from 1990 was greater than the total LUCAS CWD carbon. This meant CWD carbon estimates for 1990 in these plots were negative (Appendix 3). These calculated negative values were expected, and result from NVS trees falling in part out of the plot. Positive errors, caused by trees rooted outside a plot falling in part into the plot, were also expected. Because these two sources of error will on average cancel out, it is important to include any negative plot carbon stock estimates when calculating an overall average to ensure it is unbiased.

Overall, the distribution of CWD carbon between plots is strongly positively skewed with a few plots having very high stocks, while the majority having much lower stocks (Fig. 4). There is, however, a strong positive correlation ($r=0.87$) between the LUCAS and the 1990 stock estimates.

Figure 4 Plot-level CWD carbon stocks estimated for 1990 versus the LUCAS year (2002–2006) estimates



4.4.2 Uncertainty and sensitivity analysis

Table 5 gives the estimated CWD carbon stocks and stock changes averaged across the 31 plots in the study with 95% confidence intervals based on the plot-to-plot variation. Also shown are estimated CWD carbon stocks in 1980 and the average annual change since 1980. On average, CWD carbon increased from 1990 by 0.22 t/ha/year across the 31 NVS plots. However, this small annual increase does not differ significantly from zero, indicating there is no clear evidence for an overall increase in CWD carbon over the period.

The 95% confidence intervals given in Table 5 are based purely on plot-to-plot variability and take no account of any biases in the underlying models used to obtain the carbon stock and stock change estimates. The effects of varying various important model parameters on carbon change estimates are shown in Table 6, ordered from most sensitive to least sensitive. The stock change estimate was particularly sensitive to the first four parameters, and relatively insensitive to the last two. However, none of the parameter changes would alter the picture substantially, and the conclusion of

no significant changes in CWD between 1990 and the LUCAS measurement dates would hold even with altered parameter values.

Table 5 Estimated CWD carbon stocks in 1980, 1990, and the LUCAS year (2002-2006) and stock changes, averaged across all plots with 95% confidence intervals

Variable	Mean and 95% confidence interval
1980 CWD carbon stock (t/ha)	20.2 ± 14.2
1990 CWD carbon stock (t/ha)	21.6 ± 10.7
LUCAS CWD carbon stock (t/ha)	24.6 ± 9.6
1990–LUCAS carbon stock change (t/ha)	3.0 ± 5.4
1980–LUCAS annual carbon stock change (t/ha/yr)	0.191 ± 0.349
1990–LUCAS annual carbon stock change (t/ha/yr)	0.218 ± 0.369

Table 6 Estimated annual CWD carbon stock change estimates (t/ha/yr) that result when model components are varied by ±10% and ±25% from their assumed values

Component	–25%	–10%	+10%	+25%
CWD wood density	0.38	0.28	0.15	0.05
CWD Volume	0.38	0.28	0.15	0.05
CWD decay class modifiers	0.32	0.26	0.18	0.11
Decay Constant	0.37	0.28	0.15	0.05
Wood density	0.17	0.20	0.24	0.27
Volume	0.16	0.20	0.24	0.27

5. Mitigation opportunities for indigenous forest and shrublands

5.1 Introduction

The purpose of this section of the report is to provide an estimate of the area and carbon sequestration potential of indigenous forest that could be established on pastoral lands that are at significant risk of erosion—that is, on marginal pastoral lands. Such forests would be Kyoto-compliant, and would also provide substantial environmental co-benefits. An earlier analysis for MAF reported on areas of Crown-owned lands that are potentially available for indigenous (and exotic) forestry, split into total and marginal lands by North and South Islands (Sutherland et al. 2006).

In this report, we extend the former analysis in four ways:

- The analysis includes privately owned, as well as Crown-owned, lands.
- A more conservative approach is taken to estimating available land area, by first removing areas recorded as woody vegetation in LUCAS Ecosat imagery from areas recorded as grassland in the LCDB2 database.
- An alternative, more conservative, definition of marginal land is used (i.e. the resultant lands are more “marginal”) than the original definition—with a reference to the original definition retained for comparison.
- The analysis provides a breakdown of land ownership area by (Regional Council) region.

5.2 Analysis, Results and Discussion

5.2.1 Generation of activity data

For this report, lands with the potential to establish Kyoto forests (Kyoto-compliant lands) are defined as:

- areas classified as grasslands in the 2nd edition of the NZ Landcover Database (LCDB2). These grasslands comprise the following: high-producing grassland; low-producing grassland; tall tussock grassland; and depleted grassland (i.e. grassland classes 40–44 in LCDB2).
- areas classified as having woody cover in LUCAS Ecosat data as at c. 2000 are excluded from that compilation.

Marginal lands are then defined as resultant grasslands on Class 7 or 8 land, plus:

- either: grassland areas on Class 6 land that has an erosion potential rating (2–5) of moderate to extreme (less conservative, i.e. more Kyoto-compliant lands)
- or: grassland areas on Class 6 land that has an erosion potential rating (3–5) of severe to extreme (more conservative, i.e. less Kyoto-compliant lands)

The erosion potential rating was taken from the New Zealand Land Resource Inventory (NZLRI; Eyles 1985).

The area of Kyoto-compliant marginal lands was then split by region (Regional Council boundaries) and by the following ownership classes:

- Crown lands—dataset supplied by the Department of Conservation (DOC), updated from an earlier analysis to remove most areas that are apparently under the control of multiple agencies.
- Private lands—that is, all non-Crown lands.
- Maori lands—proprietary dataset developed by Landcare Research from data held by Te Puni Kokiri.

Considerable areas of Kyoto-compliant lands identified by the above analysis is not climatically suitable for tree growth—particularly the higher montane tussock areas owned by DOC. A further limitation was therefore placed on potentially available lands: for indigenous forests, the limit of growth was taken as the temperature at which the “tree-line” occurs. In New Zealand, the tree-line for indigenous forest commonly occurs where the mean monthly temperature of the warmest month is about 10°C (Wardle 1985). Mean monthly temperature was taken from the Land Environments of New Zealand database (LENZ; Leathwick et al. 2002).

The results of the above analysis are given in Table 7 and show that there are about 4.6 Mha of marginal land available for Kyoto-compliant indigenous forestry, or about 2.7 Mha if the more conservative definition of marginal land is adopted. Note that the values in Table 6 for the area of Crown-owned marginal lands are about 150 000 ha lower than those reported earlier (Sutherland et al. 2006), primarily in the South Island. This is because the earlier analysis was based on area mapped as grassland in the LCDB2 dataset alone. In this analysis, LCDB2 grassland areas were considered ineligible for establishing Kyoto forests if woody (shrub) vegetation, as recorded in the LUCAS dataset, was present. The analysis reported here is therefore somewhat conservative, as not all woody vegetation recorded in the LUCAS dataset will have the potential to reach forest proportions.

Table 7 Marginal lands available for indigenous forestation. For each region, two entries are shown: the upper includes Class 6 lands with erosion potential ratings of moderate to extreme, and the lower Class 6 lands with erosion potential ratings of severe to extreme. Both entries include all Class 7 and 8 lands

Region	Crown Land (ha)	Private Lands (ha)		Total
		Total	Maori-owned	
Northland	13 530	185 689	7808	199 219
	10 666	106 685	7808	117 351
Auckland	1673	45 150	140	46 823
	327	11 798	140	12 125
Bay of Plenty	4564	88 053	9741	92 617
	3617	21 648	9741	25 265
Waikato	24 637	319 264	13 388	343 901
	13 122	106 134	13 388	119 256
Taranaki	2510	90 344	388	92 854
	1609	29 646	97	31 255
Gisborne	4512	285 522	39 367	290 034
	1983	141 548	39 367	143 531
Manawatu-Wanganui	42 942	644 783	14 180	687 725
	33 996	236 402	14 180	270 398
Hawke's Bay	16 994	357 311	23 356	374 305
	8904	155 416	15 930	164 320
Wellington	6833	357 311	957	364 144
	5215	64 988	403	70 203
Nelson-Marlborough	207 780	47 588	44	255 368
	193 891	28 715	37	222 606
Westland	184 563	11 183	29	195 746
	180 922	7557	27	188 479
Canterbury	646 032	145 225	29	791 257
	555 366	78 706	13	634 072
Otago	475 901	219 381	64	695 282
	402 470	131 945	7	534 415
Southland	177 610	39 139	172	216 749
	156 109	8580	130	164 689
Total	1 810 081	2 835 943	109 663	4 646 024
	1 568 197	1 129 768	101 268	2 697 965

5.2.2 Modelling of indigenous forest and shrubland sequestration rates

Estimation of mean carbon sequestration rates for indigenous forest is more difficult than for exotic forests. There are two reasons for this. First, a systematic inventory of New Zealand's indigenous forests has not yet been completed. At least two cycles of inventory would be required to calculate mean growth rates. Second, even if such data could be obtained, the empirical data are likely to have a strong age-class bias. Across the whole native-forest estate, carbon sequestration rates are likely to be strongly dominated by the large proportion of relatively old-growth forests, and so be less relevant for new plantings and/or forest regeneration. For new plantings and regeneration, it is necessary to obtain growth rates over the first few decades of newly established stands.

For estimation of carbon sequestration in new indigenous forests, it is therefore necessary to turn to model-based approaches.

The most comprehensive information on rates of biomass growth, and therefore carbon sequestration, in indigenous forests comes from modelling work published by Hall and Hollinger (2000), Hall et al. (2001), and Hall and McGlone (2006). These papers used a forest-growth and nutrient dynamics model extensively calibrated against plot-measured data from the National Vegetation Survey (NVS). Using this model, forest succession and biomass were estimated over a 2000-year period for 10 regionally representative sites from Auckland to Otago (Hall & Hollinger 2000). For the sites modelled, carbon was sequestered at rates between $0.81 \text{ t C ha}^{-1}\text{yr}^{-1}$ ($2.0 \text{ t CO}_2 \text{ ha}^{-1}\text{yr}^{-1}$) and $1.88 \text{ t C ha}^{-1}\text{yr}^{-1}$ ($6.9 \text{ t CO}_2 \text{ ha}^{-1}\text{yr}^{-1}$) during the first 100 years, with gradually declining rates thereafter. Maximum carbon stock estimates varied from 440 t C ha^{-1} ($1600 \text{ t CO}_2 \text{ ha}^{-1}$) for undisturbed kauri/tawa/podocarp forest in Northland, to 160 t C ha^{-1} ($580 \text{ t CO}_2 \text{ ha}^{-1}$) for Hall's totara/mountain beech/mountain toatoa forest at Twizel. The time taken to reach maximum carbon stocks was also highly variable, ranging from 155 years at Craigieburn (inland Canterbury) and Glendhu (south Otago), to 1000 years at Reefton. That is, initial sequestration rates, maximum carbon stocks and time to reach maximum carbon stocks all varied strongly as functions of both climate and species.

Weighting the modelled carbon-accumulation rates for the different species by their areal distribution—which is strongly beech-dominated—results in a national mean indigenous forest carbon sequestration rate of $1.4 \text{ t C ha}^{-1}\text{yr}^{-1}$ ($5 \text{ t CO}_2 \text{ ha}^{-1}\text{yr}^{-1}$) during the first 100 years of stand growth. This sequestration rate decreases to $0.95 \text{ t C ha}^{-1}\text{yr}^{-1}$ ($3.5 \text{ t CO}_2 \text{ ha}^{-1}\text{yr}^{-1}$) over a 200-year average and to $0.62 \text{ t C ha}^{-1}\text{yr}^{-1}$ ($2.3 \text{ t CO}_2 \text{ ha}^{-1}\text{yr}^{-1}$) over 300 years. Although rates of carbon sequestration for indigenous forests are considerably lower than those for exotic forests, indigenous forests can continue to build carbon stocks over a much longer period, while exotic forests generally pass through a series of harvesting/replanting cycles. Depending on location, indigenous forests may deliver larger permanent carbon stocks than exotic forests in the medium to long term (i.e. more than c. 100 years).

Instead of planting, indigenous forest may be established in many areas through natural regeneration. Natural succession typically begins with grassland areas reverting to shrubland and later being replaced by species characteristic of the mature forest community. For shrubland regeneration, from the time that full canopy cover is achieved after about 10 years, higher rates of carbon sequestration can be expected than for indigenous-forest growth, until shrubland maturity—that is, during the first 35–50 years (depending on site conditions). National mean rates of carbon sequestration in manuka/kanuka shrubland are about $2.2 \text{ t C ha}^{-1}\text{yr}^{-1}$ ($8 \text{ t CO}_2 \text{ ha}^{-1}\text{yr}^{-1}$), and rates twice as high can be achieved under more favourable climates (Trotter et al. 2005). However, once the shrubland stands reach maturity, the rate of carbon storage slows substantially, and little further net carbon is stored during an intervening period until emergent indigenous forest species begin to replace the shrubland canopy. This intervening period with little additional carbon storage in effect lowers the long-term average rate of carbon sequestration within a manuka/kanuka/indigenous-forest successional sequence. For the purposes of calculations here we therefore consider lands reverting to manuka/kanuka shrubland to have the same overall long-term rates of carbon sequestration as indigenous forest.

5.3 Mitigation Opportunities

If the areas of Kyoto-compliant marginal grasslands suitable for indigenous forestry given in Table 7 are multiplied by the 100-year nationally averaged sequestration rate, a first estimate can be obtained of the potential carbon sequestration rate by region. These results are shown in Table 8, and suggest that a national carbon sequestration rates of about $14\text{--}24 \text{ Mt CO}_2 \text{ yr}^{-1}$ could be achieved. This table should be interpreted cautiously, as application of a single national average carbon sequestration rate is ultimately not an overly reliable predictor of sequestration rates on a

regional basis—with rates in the drier and colder regions being significantly below those shown, and vice versa (although the national average rate is weighted by beech forest, which is largely located in colder though wetter areas than marginal pasture lands). Nonetheless, using the figures in Table 8 to give an approximate ranking of regions by likely importance reveals some unexpected trends: for example, the Manawatu-Wanganui and Waikato regions rank ahead of Gisborne in terms of sequestration potential (although perhaps not in terms of environmental priority).

Table 8 Estimates of carbon accumulation on marginal lands available for indigenous forestation. For each region, two entries are shown: the upper includes Class 6 lands with erosion potential ratings of moderate to extreme, and the lower Class 6 lands with erosion potential ratings of severe to extreme. Both entries include all Class 7 and 8 lands. For calculating sequestration rates, it was assumed that all areas have achieved 100% crown cover. All data have been calculated using a single national average carbon sequestration rate

Region	Crown Land (Mt CO ₂ yr ⁻¹)	Private Lands (Mt CO ₂ yr ⁻¹)	Total (Mt CO ₂ yr ⁻¹)
Northland	0.07	0.95	1.02
	0.05	0.55	0.60
Auckland	0.01	0.23	0.24
	0.00	0.06	0.06
Bay of Plenty	0.02	0.45	0.48
	0.02	0.11	0.13
Waikato	0.13	1.64	1.77
	0.07	0.54	0.61
Taranaki	0.01	0.46	0.48
	0.01	0.15	0.16
Gisborne	0.02	1.47	1.49
	0.01	0.73	0.74
Manawatu-Wanganui	0.22	3.31	3.53
	0.17	1.21	1.39
Hawke's Bay	0.09	1.83	1.92
	0.05	0.80	0.84
Wellington	0.04	1.83	1.87
	0.03	0.33	0.36
Nelson-Marlborough	1.07	0.24	1.31
	1.00	0.15	1.14
Westland	0.95	0.06	1.00
	0.93	0.04	0.97
Canterbury	3.32	0.75	4.06
	2.85	0.40	3.25
Otago	2.44	1.13	3.57
	2.07	0.68	2.74
Southland	0.91	0.20	1.11
	0.80	0.04	0.85
Total	9.29	14.56	23.85
	8.05	5.80	13.85

6. Current best estimates of indigenous forest/shrubland carbon stocks and change

6.1 Mitigation Opportunities for Indigenous Forest and Shrubland

6.1.1 Stocks and change under current land use/management (1990–2020)

Current evidence indicates that under present management, live (Chapter 4.3) and dead biomass (Chapter 4.4) stocks in indigenous forests are at steady-state. This is strongly indicative that soil carbon stocks in these forests can also be expected to be at steady-state, with inputs equalling decomposition rates.

The largest threat to the carbon balance of existing indigenous forests is likely to be the process of climate change itself. Increased temperatures are likely to result in increasing decomposition rates in the soil organic matter and dead organic matter (litter, and coarse woody debris) pools. However, such increases may be offset by increasing forest growth rates, and thus inputs to these pools, since growth in significant areas of existing indigenous forest will be temperature-limited. By contrast, acting both against increases in the live (and dead) biomass pools from increased growth, and more rapid decomposition in the soil organic matter and dead organic matter pools, with increasing temperature would be any corresponding decreases in rainfall. Overall, provided temperatures do not advance to the point where species viability is threatened, it seems probable carbon stocks in indigenous forests are not likely to change significantly under current management between now and 2020, or perhaps out to about 2050.

6.1.2 Stocks and change for post-2012 mitigation options

New Zealand presently has a large area of marginal pasture lands that would benefit substantially from the erosion control afforded by an indigenous forest cover—between 2.7 and 4.6 Mha (Table 7). If indigenous forests were established on all available marginal lands, they would store about 6–10 Mt CO₂ yr⁻¹ between 2012 and 2020, assuming that it takes on average 10 years to achieve 100% crown cover under either natural or assisted regeneration. To put this in perspective, New Zealand's present obligations under the Kyoto Protocol require net offsets of about 7.3 Mt CO₂ yr⁻¹. Clearly, indigenous afforestation/reforestation has considerable potential to help meet New Zealand's future obligations through emissions offsets, while also offering substantial environmental and biodiversity benefits and having little long-term impact on primary pasture-based production. That said, there would also be significant costs associated with a widespread indigenous forestation programme. These include pest-control costs and costs for establishing forest in those areas without suitable seed sources.

Although rates of carbon sequestration for indigenous forest are considerably lower than those for exotic forest, indigenous forest continues to build carbon stocks over a much longer period, while exotic forest generally passes through a series of harvesting/replanting cycles. Depending on location, indigenous forest may therefore deliver a larger permanent carbon stock than exotic forest in the medium- to long-term (i.e. more than c. 100 years).

6.1.3 Comparison with other recent studies on indigenous forest mitigation potential

For completeness, we summarise here work recently completed to assess the potential to increase carbon stocks in the Conservation estate, some 8 Mha. The work is presented in Carswell et al. (2008), and the overall conclusions are reproduced here with the permission of the Department of Conservation. A more spatially detailed approach to forecasting carbon stocks was taken than used in the remainder of this report, and the increase in stocks was based on succession to the forests

predicted to occur by the Land Environments of New Zealand (LENZ) dataset (Leathwick 2001). This more locally based approach to estimating carbon sequestration could be considered in the future as a way to move from the nationally averaged rates of indigenous forest carbon sequestration used elsewhere in this report to regionally averaged (or finer scale) rates.

The current carbon stock of the c. 8 Mha of conservation land in New Zealand is c. 2400 Mt C or 8800 Mt CO₂. Indigenous forests contain 66% of the carbon within conservation land, and indigenous forest-shrubland/grassland vegetation contains an additional 11%. The single largest stock of carbon on conservation land currently resides in beech forest, which contains 635 Mt C—27% of total carbon stock.

Potential carbon storage on conservation land was predicted from potential vegetation cover. The estimates of current and potential carbon stocks are, however, based on some untested assumptions. In addition, predicting the potential carbon stock is particularly difficult because of the absence of empirical data on actual rates of carbon accumulation during the process of succession from forest-shrubland to forest. A detailed analysis of the current vegetation types suggests that the carbon stock of conservation land could potentially reach 2586 Mt C or 9490 Mt CO₂—an 8% increase on current carbon stocks, largely through an increase in the areas of ‘lowland podocarp–broadleaved forest’, ‘highland podocarp–broadleaved forest’ and ‘beech forest’. Most of the increase would occur through the completion of successions of existing seral vegetation, i.e. of shrubland–forest or grassland–shrubland to forest, over periods ranging from a few decades to as long as 300 years.

The CO₂ estimated to potentially be sequestered through afforestation/reforestation (A/R) of the 400 000 ha of conservation land (5% of the total) considered to have been ‘non-forest’ as at 31 December 1989 amounts to about 268 Mt CO₂. Carbon gains are certain to occur in favourable ‘non-forest’ areas through relatively inexpensive management actions such as exclusion of domestic stock and low-level wild-animal control. On sites favourable for low-intensity management of natural regeneration, we predict significant gains through afforestation could be largely achieved in 100 years. Transitions to ‘Kyoto forest’ shrublands could occur in as little as 10 years, providing a modest sequestration rate over a relatively short period. The afforestation of drylands, however, is likely to occur over centuries rather than decades.

The best ways that conservation land can be managed to help New Zealand meet its obligations under Article 4.1(d) of the United Nations Framework Convention on Climate Change to “*promote and cooperate in the conservation and enhancement, as appropriate, of sinks and reservoirs of all greenhouse gases*” (UNFCCC 1994) are to: (a) minimise losses of carbon from conservation land (e.g., through prevention of forest fire); and (b) increase carbon stocks by establishing new forests, particularly through afforestation of grasslands. Unlike gains in carbon from management of existing forests, A/R gains per hectare are relatively large and fairly easy to quantify over short-time periods (5–10 years).

The biggest potential risk to carbon already stored on conservation land is natural disturbance (particularly through erosion, earthquakes and volcanism). The legacies of repeated burning and grazing and also logging now constrain potential forest composition and therefore carbon storage.

Large uncertainty remains about the potential impacts of climate change on New Zealand’s forests, although increased drought in eastern areas may infer increased frequency of wildfire, which would have a significant impact on carbon storage in a flora largely unadapted to fire. Because of nitrogen limitation in many forests and low rates of nitrogen deposition, there are unlikely to be substantial gains in carbon storage as a result of increased atmospheric CO₂. However, there is considerable potential for loss of soil carbon if predicted temperature increases stimulate respiration as some studies predict. Increasing drought in eastern parts of the country is also predicted to reduce carbon uptake, while gains are possible in western regions where temperatures will likely rise along with rainfall.

No significant risks to carbon stocks as a result of exotic insect invasion were discovered. Evidence suggests indigenous forest species, particularly *Nothofagus* spp., are relatively resistant to exotic insect attack but much more work is required on other genera of forest trees and on the potential for ant species, in particular, to impact successional processes.

6.2 Discussion

6.2.1 Implications of forecasts and scenarios

The analysis of available land in different erosion-risk categories and currently covered by grassland has shown that New Zealand has about 4.6 Mha of marginal, erosion-prone pasture lands that could potentially be used for establishing Kyoto-compliant indigenous forest (Table 7). Even if a more conservative definition of marginal land is adopted (i.e. lands with ‘moderate’ erosion risk omitted), 2.7 Mha are still considered as suitable. Of this land, about 1.8 Mha (or 1.6 Mha under the more conservative categorisation) are in Crown ownership (including a small proportion of lands under QEII Covenants) and 2.8 Mha (1.1 Mha) are in private ownership. Of the privately owned lands, about 4% (or 9%) are owned by Maori, with most of these lands tending to be in the more marginal class 7 and 8 categories.

To estimate the potential for carbon sequestration on these lands by indigenous forestation only, we have used nationally averaged sequestration rates modelled for existing forested lands (Section 6.1.2, above). However, it is likely that forestation scenarios involving a mix of exotic and indigenous forest establishment are more likely, as landowners seek to maximise economic gain.

In Table 9, we present what is considered to be a likely scenario for the potential of forests to achieve emissions offsets. This scenario assumes that exotic forest is planted on all suitable marginal lands, with indigenous forest confined to lands not suited to exotic forest. By way of comparison, we include also the all-indigenous forest scenario developed above in Section 6.1.2, and the all-exotic (on all suitable marginal lands) developed earlier in Chapter 3.

Table 9 Scenarios for carbon sequestration by forests on Crown- and privately-owned Kyoto-compliant marginal lands. For exotic forests, suitable lands were taken as those with mean annual temperatures $\geq 8^{\circ}\text{C}$, whereas for indigenous forest, suitable lands were taken as those up to the present tree-line. The rates used for the exotic forest component in the table below have been reduced by 25% for plantings on Crown-owned lands (i.e. from a current national mean value of $29 \text{ t CO}_2 \text{ yr}^{-1}$ to $22 \text{ t CO}_2 \text{ yr}^{-1}$), to account for the fact that average Crown-owned lands have harsher climatic conditions than existing exotic forested lands.

Scenario (for the more conservative marginal land case)	Carbon storage (Mt CO ₂ yr ⁻¹)
1. Indigenous forests, all suitable marginal Crown-owned lands	8
2. Indigenous forests, all suitable marginal privately owned lands	6
3. Exotic forests, all suitable marginal Crown-owned lands	4
4. Exotic forests, all suitable marginal privately owned lands	27
5. Exotic forest on all suitable marginal Crown-owned lands, indigenous forest on remaining suitable marginal Crown-owned lands	11
6. Exotic forest on all suitable marginal privately owned lands, indigenous forest on remaining suitable marginal privately owned lands	28

Irrespective of the scenario, it is clear forestation of marginal lands provides useful gains in emissions offsets, and a major forest establishment programme involving long-lived exotics on suitable lands could offer New Zealand an effective approach to meeting a greater share of commitments to emissions reductions while enhancing sustainable land use. As noted earlier, New Zealand requires offsets of about 7.3 Mt CO₂ yr⁻¹ during the first commitment period to meet its international obligations under the Kyoto Protocol.

6.2.2 Effects of information gaps/uncertainties on forecast/scenario reliability

The overall reliability of forecasts of net removals by indigenous forests is likely to be reasonably high at national scales, although, as one might expect, the absolute area available for mitigation strategies varies considerably with the particular definition of “marginal land” adopted. At a regional scale, the same issues arise with the marginal land definition as at the national scale. However, at regional scales a further uncertainty arises: lack of regionally-determined carbon sequestration rates. Several steps are necessary to remove this uncertainty:

- As an interim step, regional carbon sequestration rates could be forecast using the same approach recently used for analysis of the Conservation estate. That is, by using LENZ data on potential forest cover as representing the climax forest type, and modelling sequestration rates through the vegetation succession that results in that climax type. This process does, however, have significant weakness in terms of the likely accuracy of predicted carbon sequestration rates through the early establishment phase relevant to new forestry projects over the next few commitment periods.
- To improve accuracy of predicted carbon sequestration rates especially in the early phases of indigenous forest establishment, a substantial investment in model development and validation for young forests would be required.

7. Conclusions and recommendations

7.1 Present Status of Studies, Datasets, Analyses and Forecasts

Biomass in a reasonable number of indigenous forests has now been measured in repeat measurements to allow an assessment of changes over time. Analysis of these existing data has shown a small, but not statistically significant, increase over time. Preliminary data for the dead wood pool also indicate that temporal changes are not significant, although a larger study is yet required before this can be fully confirmed. Any expanded study on dead-wood may have to be model-based, however, as there are relatively few sites with a good temporal record of dead-wood stocks.

In terms of mitigation opportunities, sufficient data and analyses are available to indicate a significant potential for indigenous forestation of marginal land to create very useful levels of emissions offsets in the medium term. Increasing forest cover on marginal lands would also offer substantial environmental benefits, including erosion control and enhancement of indigenous biodiversity. To further refine mitigation scenarios for re-establishment of indigenous forests, a better definition of exactly what land is considered “marginal” would be required. This should be combined with an analysis of the effect of carbon prices in driving conversion of marginal pasture lands to “carbon farming”.

7.2 Information Gaps, Uncertainties, and Research Priorities

To better characterise the impact of indigenous forests on New Zealand's future net position, and to characterise their potential for emissions mitigation, the following need to be considered:

- Obtaining a more precise definition of marginal pasture land, and of the carbon price at which “carbon farming” on such land becomes a viable economic proposition. This should be obtained by region and probably by land classes within regions.
- Developing models of regional, and preferably sub-regional, rates of carbon sequestration in indigenous forests based on likely successional pathways.
- Investing substantially in developing and validating models of establishment and growth of indigenous shrublands and forests, including of rates of canopy closure under natural regeneration regimes.
- Researching and defining land-management practices that can enhance natural regeneration rates of indigenous forest, and encourage rapid succession from lower-biomass shrubland to higher-biomass tall forest.
- Development of faster growing, high-value indigenous timber species for establishing viable long-rotation managed forests.

7.3 Implications of Accounting and Mitigation Options for New Zealand's Post-2012 Net Position

If, as appears very likely, New Zealand's existing indigenous forests are at a carbon steady-state, they will have no impact on New Zealand's post-2012 position unless both accounting under Article 3.4 (or equivalently some form of net-net accounting) becomes mandatory and the forests are also significantly affected by fire, earthquake, volcanism or disease. Some minor increases in the live biomass pool may occur due to rising temperature (since growth is temperature-limited in many places), but these are likely to be offset by small losses in dead-wood stocks and soil mineral carbon. There could be losses if the forests experience drier conditions, although most indigenous forests are located in areas of more than adequate rainfall. Rising CO₂ levels are unlikely to stimulate growth, as New Zealand's indigenous forests are generally strongly nutrient limited.

There are significant opportunities for emissions mitigation using indigenous forest schemes, particularly when used to convert erodible pasture lands to a more sustainable land use. Realistically, however, only the most marginal lands are likely to be converted to indigenous forests, while slightly better land might be used for exotic forest plantings. Total carbon sequestration on both private and Crown-owned marginal lands could amount to between 6 and 28 Mt CO₂ yr⁻¹, depending on the mix of indigenous and exotic afforestation (see Section 6.2.1.).

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Appendix 1

Estimated CWD carbon at the LUCAS measurement for each plot.

Plot	Measurement year	Carbon (t/ha)		
		Spars	Logs	Total CWD
AJ133	2006	126.4	3.6	130.0
AJ134	2006	3.1	28.1	31.1
AK134	2006	8.0	16.1	24.2
AW124	2005	24.3	7.9	32.2
AX124	2005	32.2	47.2	79.5
BD121	2004	1.4	2.5	4.0
BG121	2003	0.5	28.5	29.0
BN109	2002	13.1	0.6	13.7
CB74	2002	0.0	9.6	9.6
CL55	2004	9.6	17.0	26.6
CM104	2002	1.4	3.7	5.1
CM55	2005	0.3	7.5	7.7
CM56	2002	0.5	36.8	37.3
CM97	2002	5.3	3.0	8.4
CN97	2002	3.5	10.2	13.7
CQ60	2005	0.0	13.4	13.4
CQ61	2004	0.3	1.6	1.9
CR61	2004	0.0	0.0	0.0
CR62	2004	22.4	3.6	25.9
CZ74	2004	2.5	7.9	10.4
DA57	2003	0.0	13.7	13.7
DA58	2004	0.0	22.0	22.0
H158	2004	12.6	6.3	19.0
H159	2006	1.5	4.2	5.7
N155	2002	4.4	15.3	19.7
N156	2006	10.6	53.2	63.8
N157	2006	0.0	18.8	18.8
N158	2005	0.0	38.2	38.2
P158	2002	3.6	31.0	34.6
U153	2005	19.6	0.8	20.4
V182	2004	0.0	3.2	3.2
Mean	2004	9.9	14.7	24.6

Appendix 2

Estimated above ground carbon in trees that died between 1990 and the LUCAS measurement date for each plot.

Plot	Dead tree count (1990–LUCAS)	Total Carbon in Dead Trees (t/ha)	
		At time of mortality	At LUCAS measurement
AJ133	6	60.2	50.2
AJ134	1	1.9	1.6
AK134	2	4.8	3.9
AW124	5	18.2	16.2
AX124	1	0.2	0.2
BD121	0	0.0	0.0
BG121	2	19.9	16.2
BN109	4	17.5	14.1
CB74	0	0.0	0.0
CL55	4	17.9	16.6
CM104	1	4.6	3.9
CM55	2	2.0	1.9
CM56	3	17.2	14.0
CM97	3	11.1	9.5
CN97	1	4.0	3.1
CQ60	0	0.0	0.0
CQ61	2	1.3	1.2
CR61	0	0.0	0.0
CR62	0	0.0	0.0
CZ74	2	4.7	4.2
DA57	3	2.6	2.5
DA58	3	30.7	29.0
H158	4	32.9	27.4
H159	1	0.5	0.4
N155	1	0.5	0.4
N156	12	61.9	48.1
N157	0	0.0	0.0
N158	0	0.0	0.0
P158	0	0.0	0.0
U153	2	22.9	19.1
V182	0	0.0	0.0
Mean	2.1	10.9	9.1

Appendix 3

Estimated CWD carbon stocks in 1990 and at the LUCAS measurement, and estimated annual carbon stock change for each plot.

Plot	1990 CWD carbon (t/ha)	LUCAS CWD carbon (t/ha)	Annual CWD change (t/ha/yr)
AJ133	115.2	130.0	0.926
AJ134	42.6	31.1	-0.717
AK134	29.2	24.2	-0.318
AW124	22.5	32.2	0.646
AX124	111.8	79.5	-2.156
BD121	5.5	4.0	-0.107
BG121	17.3	29.0	0.904
BN109	-0.5	13.7	1.184
CB74	12.6	9.6	-0.253
CL55	13.8	26.6	0.916
CM104	1.6	5.1	0.290
CM55	8.3	7.7	-0.035
CM56	30.7	37.3	0.554
CM97	-1.5	8.4	0.823
CN97	14.0	13.7	-0.026
CQ60	18.9	13.4	-0.366
CQ61	1.0	1.9	0.063
CR61	0.0	0.0	-0.001
CR62	35.7	25.9	-0.700
CZ74	8.6	10.4	0.134
DA57	15.0	13.7	-0.103
DA58	-9.6	22.0	2.256
H158	-11.6	19.0	2.185
H159	7.7	5.7	-0.124
N155	25.4	19.7	-0.476
N156	22.7	63.8	2.570
N157	27.1	18.8	-0.520
N158	53.8	38.2	-1.043
P158	45.5	34.6	-0.912
U153	1.9	20.4	1.235
V182	4.4	3.2	-0.085
Mean	21.6	24.6	0.218

Appendix 4

Summary of live tree biomass C in NVS and remeasured CMS/Lucas plots.

NVS Plot identifier	NVS live tree C (t/ha)	NVS live tree stems (no./plot)	CMS plot identifier	CMS live tree C (t/ha)	CMS live tree stems (no./plot)	NVS year	CMS year	Total years	Change in C (t/ha/yr)	Change in number of live stems (no./plot/yr)
25 2	167.2	23	AB143	159.8	12	1978	2006	28	-0.266	-0.393
33 1	255.9	24	AC141	295.2	19	1978	2006	28	1.405	-0.179
3 6	434.2	89	AC160	333.5	73	1978	2006	28	-3.598	-0.571
2 8	222.9	95	AC161	249	114	1978	2004	26	1.004	0.731
37 3	228.7	114	AD141	261	106	1978	2004	26	1.245	-0.308
31 5	182.5	33	AD142	204.1	65	1978	2002	24	0.898	1.333
38 4	209.9	40	AE141	263.1	68	1978	2004	26	2.048	1.077
40 8	123	35	AE142	156.7	55	1978	2004	26	1.298	0.769
2 2	60.2	82	AE145	283	122	1973	2005	32	6.961	1.25
33 4	214.4	53	AG144	182.2	37	1980	2006	26	-1.236	-0.615
36 4	98.9	65	AG145	112.2	59	1980	2006	26	0.512	-0.231
43 5	123.5	65	AG147	154.5	56	1973	2004	31	1	-0.29
4 4	183.6	10	AG166	258.8	35	1984	2004	20	3.759	1.25
3 1	332.5	149	AJ133	600.1	192	1992	2006	14	19.113	3.071
1 15	570.6	76	AJ134	583.7	95	1992	2006	14	0.937	1.357
2 14	239.2	164	AK134	238.2	133	1992	2006	14	-0.076	-2.214
F1 47	49.5	144	AO127	56.7	236	1999	2002	3	2.417	30.667
G 2	135.4	146	AP128	144.3	162	1999	2006	7	1.273	2.286
G 4	162.2	112	AS128	152.4	108	1999	2005	6	-1.628	-0.667
D 6	251.6	151	AU125	249.5	175	1999	2006	7	-0.296	3.429
D 7	385.9	86	AV125	466.6	93	1999	2004	5	16.13	1.4
B 8	671.9	233	AW122	569.2	265	1999	2006	7	-14.676	4.571
C 8	103.1	109	AW123	182.4	182	1999	2004	5	15.843	14.6
69 1	339.9	57	AW124	272.4	192	1996	2005	9	-7.498	15
70 3	29.6	117	AX124	145.3	478	1996	2005	9	12.858	40.111
BE CT	290.8	56	AY112	483	88	1989	2005	16	12.01	2
18 3	60.4	41	AZ133	102.9	197	1978	2004	26	1.637	6
4 8	1049.5	76	BA121	823.6	92	1992	2004	12	-18.829	1.333
7 8	279.5	26	BA122	264.9	103	1993	2004	11	-1.323	7
D 11	147.7	117	BA125	156.9	129	1999	2005	6	1.53	2
28 1	233.8	165	BA132	232.2	135	1978	2003	25	-0.065	-1.2
D 12	83	99	BB124	104.3	177	1999	2005	6	3.549	13
8 2	85.5	27	BC130	76.1	284	1976	2005	29	-0.324	8.862
1 1	244.5	110	BD121	289	78	1999	2004	5	8.905	-6.4
6 1	122.8	87	BE110	149.7	125	1976	2004	28	0.959	1.357
17 1	84.4	81	BE111	147.3	65	1976	2003	27	2.33	-0.593

NVS Plot identifier	NVS live tree C (t/ha)	NVS live tree stems (no./plot)	CMS plot identifier	CMS live tree C (t/ha)	CMS live tree stems (no./plot)	NVS year	CMS year	Total years	Change in C (t/ha/yr)	Change in number of live stems (no./plot/yr)
12 4	231.2	74	BF120	309.6	63	1975	2003	28	2.8	-0.393
A 16	152.6	94	BF121	159.8	81	1999	2003	4	1.806	-3.25
F 16	105.7	61	BF127	119	62	1999	2003	4	3.324	0.25
G 16	65.4	114	BF128	70.4	401	1999	2004	5	1.004	57.4
14 3	148.4	35	BG118	198.4	23	1975	2005	30	1.668	-0.4
15 4	123.7	46	BG121	156.5	75	1986	2003	17	1.931	1.706
35 3	161.8	115	BG123	197.3	95	1986	2003	17	2.088	-1.176
10 2P	214.8	281	BG125	254.3	302	1984	2004	20	1.975	1.05
61 3	215.9	13	BH120	291.1	45	1986	2004	18	4.179	1.778
F 18	109.4	107	BH127	228.8	173	1999	2003	4	29.863	16.5
HUR P1	123.6	176	BI104	171.2	136	1982	2005	23	2.072	-1.739
89 P3	310.3	135	BI105	330.9	134	1982	2004	22	0.936	-0.045
Unknown	263.1	75	BI106	270.8	68	1982	2005	23	0.333	-0.304
15 3	212.9	130	BI116	246.9	115	1980	2002	22	1.546	-0.682
21 2	117.7	128	BI117	120.2	192	1980	2003	23	0.111	2.783
17 3	292.9	83	BI118	373.4	117	1980	2005	25	3.221	1.36
3 4	26.2	21	BI119	130.1	35	1980	2004	24	4.328	0.583
31 3	57.4	48	BI127	66	231	1986	2004	18	0.476	10.167
17 1	261.2	52	BI97	271.3	44	1986	2006	20	0.509	-0.4
5 2	259.4	110	BI99	374.6	94	1986	2004	18	6.4	-0.889
23 3	361.3	62	BJ98	394.1	56	1986	2005	19	1.727	-0.316
32 2	262.2	106	BJ99	334.3	85	1986	2004	18	4.003	-1.167
LR1 3	156.7	143	BK100	167.9	116	1981	2004	23	0.483	-1.174
9 1	170.6	64	BK103	212.7	94	1982	2005	23	1.829	1.304
Control	137.1	33	BK105	157.2	33	2000	2002	2	10.065	0
66 1	93.2	99	BK118	79.2	51	1980	2005	25	-0.561	-1.92
68 1	89.1	59	BK126	72.4	216	1986	2002	16	-1.04	9.813
20 1	136.2	100	BK98	174.8	75	1986	2006	20	1.933	-1.25
37 4	299.6	65	BK99	327.3	49	1986	2002	16	1.729	-1
4 8	155.8	141	BL99	168.9	210	1986	2004	18	0.728	3.833
2 1	73.4	115	BM100	92.3	130	1981	2002	21	0.901	0.714
G 62	180.3	77	BM99	182.7	96	1981	2006	25	0.097	0.76
12 3	191.9	119	BN109	169.3	59	1982	2002	20	-1.133	-3
17 4	162.4	55	BO108	151.7	155	1982	2004	22	-0.484	4.545
11 4	135.4	106	BO109	147.6	80	1982	2002	20	0.61	-1.3
7 5	110	174	BO110	136.7	90	1982	2002	20	1.337	-4.2
22 3	292.3	41	BS103	349.8	35	1983	2004	21	2.74	-0.286
17 2	180	45	BS104	209.3	42	1983	2005	22	1.332	-0.136
8 6	366.4	47	BT102	322.8	45	1983	2005	22	-1.98	-0.091
40 7	189.4	101	BU100	208	113	1983	2006	23	0.807	0.522
32 2	456.4	40	BU101	582.7	43	1983	2006	23	5.491	0.13

NVS Plot identifier	NVS live tree C (t/ha)	NVS live tree stems (no./plot)	CMS plot identifier	CMS live tree C (t/ha)	CMS live tree stems (no./plot)	NVS year	CMS year	Total years	Change in C (t/ha/yr)	Change in number of live stems (no./plot/yr)
76 5	162.9	74	BU103	204.4	60	1984	2002	18	2.302	-0.778
44 2	204.3	77	BU108	276.3	56	1982	2005	23	3.131	-0.913
73 3	238.1	50	BV103	373.1	45	1984	2006	22	6.134	-0.227
80 2	276.3	31	BW102	263.2	25	1984	2002	18	-0.731	-0.333
28 1	223.2	64	BX112	255.9	91	1982	2003	21	1.558	1.286
30	212.7	66	BZ73	29.4	126	1985	2003	18	-10.184	3.333
H7 25	405.9	174	CA72	385.6	155	1985	2005	20	-1.013	-0.95
31	39.2	239	CB74	105.4	595	1985	2002	17	3.894	20.941
3	91.3	157	CJ95	122.9	127	1982	2006	24	1.318	-1.25
13 4	187.8	76	CL55	202.3	96	1999	2004	5	2.91	4
61 3	260.6	86	CL96	342.4	74	1985	2002	17	4.808	-0.706
F19	58.7	56	CM104	80.9	137	1986	2002	16	1.383	5.063
10 1	108.9	154	CM55	157.1	162	1999	2005	6	8.033	1.333
P3	48.1	34	CM56	66.4	53	1999	2002	3	6.096	6.333
64 2	432.2	166	CM96	499.9	143	1985	2005	20	3.387	-1.15
127 1	264.7	83	CM97	203.8	65	1998	2002	4	-15.216	-4.5
1 50	68.2	32	CN102	73	87	1986	2004	18	0.269	3.056
9 50	294.9	38	CN103	389.4	39	1986	2004	18	5.251	0.056
15 10	31.2	6	CN104	31.9	116	1986	2004	18	0.038	6.111
4 1	8.7	26	CN42	93.5	245	1988	2004	16	5.295	13.688
144 2	199.1	135	CN94	248.1	106	1985	2004	19	2.579	-1.526
88 4	361.8	102	CN97	337.9	83	1998	2002	4	-5.976	-4.75
19 1	150.5	82	CO93	169	103	1985	2005	20	0.927	1.05
52 1	253.7	98	CO95	263.2	78	1985	2005	20	0.472	-1
72 2	301.3	68	CO96	149.9	248	1985	2006	21	-7.207	8.571
96 2	203.7	113	CO97	376.1	87	1985	2004	19	9.072	-1.368
161 1	165	136	CP92	191.5	142	1985	2005	20	1.327	0.3
103 1	413.9	82	CP94	485.9	95	1985	2005	20	3.599	0.65
54 4	294.4	141	CP95	299.4	122	1985	2006	21	0.241	-0.905
24 3	60.7	57	CQ60	128.2	99	1993	2005	12	5.625	3.5
72 3	88.6	255	CQ61	92.8	291	1993	2004	11	0.383	3.273
HU CT2	195.8	186	CQ65	150.9	250	1987	2002	15	-2.993	4.267
169 2	168.3	339	CQ93	220.4	263	1985	2006	21	2.482	-3.619
2	119.6	205	CR61	149.7	241	2000	2004	4	7.527	9
2 2	152.6	73	CR62	202.6	92	2000	2004	4	12.5	4.75
1 12	187.1	50	CS72	233.4	42	1986	2004	18	2.574	-0.444
1 4	144.8	104	CU51	158.4	92	1984	2002	18	0.758	-0.667
8 10	10.7	85	CV72	47.7	153	1981	2003	22	1.683	3.091
13 10	78	40	CV73	108.8	88	1981	2006	25	1.232	1.92
146 12	829.4	39	CV83	420.8	191	1983	2005	22	-18.574	6.909
12N 4	235.4	31	CW71	123.2	21	1980	2002	22	-5.103	-0.455

NVS Plot identifier	NVS live tree C (t/ha)	NVS live tree stems (no./plot)	CMS plot identifier	CMS live tree C (t/ha)	CMS live tree stems (no./plot)	NVS year	CMS year	Total years	Change in C (t/ha/yr)	Change in number of live stems (no./plot/yr)
40 10	58.7	98	CW79	202.3	73	1983	2006	23	6.243	-1.087
F 20	130.5	108	CW81	88.9	97	1983	2002	19	-2.188	-0.579
27 2	140.4	33	CX69	221.1	60	1987	2002	15	5.384	1.8
81 8	244.5	154	CX78	183.6	76	1983	2006	23	-2.647	-3.391
1	126.6	97	CY54	159.6	122	2000	2002	2	16.509	12.5
119 2	108.2	31	CY71	183	47	2000	2005	5	14.974	3.2
166 002	224.4	41	CY72	113.9	52	2000	2002	2	-55.25	5.5
7041	86	302	CY78	96.1	269	1983	2004	21	0.479	-1.571
F 33	5.4	13	CZ73	15.5	57	2001	2002	1	10.104	44
36 26	174.9	59	CZ74	158.2	90	2001	2004	3	-5.586	10.333
1 5	96.2	74	DA57	560.5	90	1992	2003	11	42.213	1.455
3A 1	71.7	53	DA58	144.2	70	1992	2004	12	6.042	1.417
44	223.7	79	DD68	114	57	1981	2002	21	-5.222	-1.048
53	294	64	DG63	368	72	1985	2002	17	4.355	0.471
41A	153.4	62	DO58	268.7	97	1982	2006	24	4.807	1.458
113A 2	106.7	184	DP55	265.6	156	1984	2004	20	7.942	-1.4
108 4	234.1	67	DP56	127.7	68	1982	2002	20	-5.319	0.05
149 D	46.1	96	DR56	78.9	83	1982	2004	22	1.492	-0.591
5 2	331.5	141	H158	322.4	122	1988	2004	16	-0.567	-1.188
6 5	140.3	55	H159	179.4	64	1988	2006	18	2.169	0.5
3 7	105.5	123	H167	129.8	125	1977	2004	27	0.899	0.074
2 7	438.6	34	H168	478.2	72	1977	2002	25	1.583	1.52
11 1	350.4	48	H170	338.1	58	1977	2004	27	-0.456	0.37
4 5	120.9	77	I167	165.2	126	1977	2004	27	1.638	1.815
4 1	356.9	73	J170	371.1	58	1997	2006	9	1.581	-1.667
7 5	139	207	J171	120.1	212	1997	2003	6	-3.156	0.833
71 3	261.6	124	K155	235.2	106	1975	2006	31	-0.852	-0.581
73 1	179.1	153	K156	192.5	124	1981	2002	21	0.636	-1.381
0 6	407.7	146	K166	316.3	121	1982	2003	21	-4.349	-1.19
11 2	250.4	87	K171	317.1	77	1997	2005	8	8.336	-1.25
78 3	133.6	65	L157	147.6	104	1975	2005	30	0.466	1.3
22 2	66.5	150	L163	91.8	150	1982	2005	23	1.099	0
12 4	314.8	100	L164	199.6	95	1982	2006	24	-4.803	-0.208
8 2	190.7	101	L169	150.9	82	1982	2004	22	-1.809	-0.864
14 1	249.6	138	L170	276.2	116	1997	2006	9	2.953	-2.444
18 1	124.9	215	L171	141.5	202	1997	2004	7	2.384	-1.857
26 1	102.4	141	M162	146.6	173	1982	2006	24	1.844	1.333
23 1	279.6	60	M163	175.7	41	1982	2005	23	-4.515	-0.826
22 3	108.2	121	M171	164.9	167	1997	2006	9	6.303	5.111
44 5	337.8	105	N155	311.8	97	1984	2002	18	-1.441	-0.444
53 1	290.4	100	N156	123.2	87	1984	2006	22	-7.596	-0.591

NVS Plot identifier	NVS live tree C (t/ha)	NVS live tree stems (no./plot)	CMS plot identifier	CMS live tree C (t/ha)	CMS live tree stems (no./plot)	NVS year	CMS year	Total years	Change in C (t/ha/yr)	Change in number of live stems (no./plot/yr)
83 1	328.1	44	N158	416.7	34	1998	2005	7	12.662	-1.429
4 19	232.8	172	O149	224.4	128	1978	2002	24	-0.352	-1.833
6 3	107.1	120	O150	124.6	114	1975	2005	30	0.581	-0.2
94 1	197.8	69	O155	232.8	63	1978	2004	26	1.345	-0.231
95 4	359.2	111	P155	326.1	158	1978	2004	26	-1.271	1.808
99 5	296.5	55	P156	151.7	44	1978	2004	26	-5.566	-0.423
68 3	81.5	70	P158	114.9	92	1998	2002	4	8.346	5.5
P 3	63	139	P169	192.5	171	1983	2004	21	6.168	1.524
A 5	344.7	30	Q146	329	77	1984	2005	21	-0.747	2.238
E 1	380.8	32	Q147	423.8	75	1984	2006	22	1.95	1.955
1 3	401.8	109	R171	432.2	81	1997	2004	7	4.348	-4
50 3	286.5	115	S181	292.3	118	1999	2004	5	1.147	0.6
5 1	186.1	111	U152	194.5	80	1999	2004	5	1.679	-6.2
13 4	285.1	88	U153	300	79	1999	2005	6	2.48	-1.5
C 23	129.9	308	U180	184.4	362	1999	2003	4	13.623	13.5
/ 8	213.6	136	U181	221	157	1999	2003	4	1.842	5.25
10 3	128.5	68	V147	109.6	67	1978	2004	26	-0.724	-0.038
PA 14	124.9	28	V182	160.6	103	1999	2004	5	7.137	15
11 2	128.1	88	W147	173.2	27	1978	2004	26	1.736	-2.346
1 2	39.6	99	X144	52.5	134	2000	2005	5	2.589	7
9 62	306.6	48	Y143	403.8	55	2000	2002	2	48.608	3.5

CHAPTER FIVE

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Summary

This Chapter deals with changes in soil carbon stocks in response to land-use change (afforestation/ reforestation/deforestation), different forest-management options and changes under cropping, pastoral agriculture, and horticulture. The chapter also discusses the role of erosion, by trying to quantify its source/sink contribution and by discussing it in the context of carbon accounting. The chapter tries both to assess the changes that need to be reported and accounted as part of the LULUCF sector under the United Nations Framework Convention on Climate Change (UNFCCC) and under the Kyoto Protocol and to identify any options for climate change mitigation in the different Sectors.

Afforestation/ Reforestation/ Deforestation

For the First Commitment Period, only soil carbon changes related to land-use change (under Article 3.3 of the Kyoto Protocol) need to be accounted. In New Zealand that principally means soils carbon changes after deforestation, the conversion of established forest to pastures or other land uses, and afforestation or reforestation, the conversion of pastures to native or exotic forests. Soil carbon is generally lost following afforestation/ reforestation with exotic forests, an important issue for New Zealand as it reduces the benefit of biomass carbon stocks in newly planted forests, and it affects large areas of the country.

Different analyses have derived estimates of soil carbon losses between 8 and 18 t C ha⁻¹ on conversion of pasture to pine forest. In particular, the wide soil sampling that had been incorporated in New Zealand's Carbon Monitoring System had developed a best estimate of a loss of soil C of 18 t C ha⁻¹, whereas analysis of paired sites had derived a lower estimate of 8.5 t C ha⁻¹. A newer analysis reported as part of this review has explicitly taken the auto-correlation between different sampling points into account, and has derived lower estimates of between 8 and 13 t C ha⁻¹. The various analyses all carry large uncertainties so that the different estimates are not statistically different from each other. Given some reasonable assumptions about the costs of carbon and the magnitude of the areas that have been reforested in New Zealand, the difference between these estimates translates into differences in the size of New Zealand's greenhouse gas liability of the order of \$150M. Additional work to further refine that estimate is considered a high priority.

As important as the magnitude of change is its time course. The current default for accounting purposes is a linear change over 20 years. It is possible, however, that the change is not linear but that the loss may be more rapid over early years after land-use change and then stabilise towards a new value, with lesser changes in later years. Such a time course would be advantageous for New Zealand as rates of plantation establishment were particularly high in the early 1990s. Under the current IPCC default, any changes in soil carbon calculated based on a linear-change assumption have to be carried through the first commitment period, whereas the liability would be substantially less if the time course is sigmoidal or exponential, and most of the change has already occurred.

Measurements of the time course of soil carbon losses are unlikely to be detailed enough to be able to provide definitive answers on the likely time course of change, and the small number of modelling studies that have addressed the question have not yet given a clear and unambiguous answer to that question. That, too, is an area where further work could be most useful in reducing New Zealand's accounting liability.

For the deforestation of exotic forests to pasture, it is presently assumed that the soil C loss incurred in a shift from pasture to forest would simply be reversed. There are very limited data to support this assumption, and the only recent New Zealand study of soil C changes after deforestation has observed very large gains in soil C. With the recent upsurge of conversions of forests to dairying, it is warranted to expend some further effort on quantifying the soil C change following deforestation.

Soil sampling to date has also deliberately excluded sites affected by recent erosion events, but since erosion is part of the experience at different sites, locations affected by erosion should, in fact, be included in any representative and carefully stratified sampling regime. It is therefore not known if the land-use factors currently used are appropriate for erodible hill-country. Carbon accounting with erosion raises other important issues that are discussed further below.

Regardless of the post-2012 accounting regime that will finally be endorsed internationally, changes in carbon stocks with afforestation/reforestation will have to be quantified and accounted. Under gross-net accounting approaches, as is currently done under Article 3.3, soil C changes partly negate the larger and opposite change in biomass C stocks. So soil C changes reduce the benefit from reforestation, and reduce the liability from deforestation.

If net-net accounting were to be adopted after 2012, the loss of soil C with reforestation would actually be advantageous as any reduced C gain would increase total 1990 baseline emissions, whereas future soil-C losses would trend towards zero as more and more forests reach an age beyond the assumed 20 years for adjustment in soil C stocks. Net-net accounting in any form would carry many other major disadvantages for New Zealand, however, so the issue of soil carbon change would be of relatively minor importance.

Biochar Amendment

The use of biochar has been proposed as a means to store more carbon in soils to improve the net greenhouse gas balance of various agricultural or forestry practices. However, to assess the full greenhouse implications of biochar addition, it is necessary to consider a range of processes and interactions at different time scales. It is, therefore, not possible to calculate the greenhouse benefit of biochar addition by simply adding the amount of biochar carbon stored to the greenhouse balance that would be obtained without the addition of biochar. In this report, we provide a mathematical framework for estimating the net carbon balance associated with the addition of biochar to cropland or pastoral soils.

Calculations performed using the framework broadly confirm that there is a substantial potential for C sequestration benefits from biochar incorporation in New Zealand soils. Nevertheless, the calculations also suggest considerable uncertainty stemming from uncertainty around some key parameters, with pessimistic calculations suggesting that over 5 years, biochar application to croplands might barely result in any net C storage, and might even be a net C source to the atmosphere for application to pastures when incorporation requires tillage. There is a wide range of values between estimates based on pessimistic and optimistic parameter settings, but in all cases it is clear that the net carbon sequestration resulting from biochar addition to soils will be considerably less than the quantity of added biochar.

This work shows the importance of introducing robust equations for accounting for biochar addition under New Zealand conditions so that major uncertainties can be identified and targeted for future research. The main uncertainties relate to the accounting of the diverted biomass used to produce biochar, the residence times of soil C and the dynamic fraction of biochar, the proportion of biochar that is effectively resistant to decomposition, and the loss of soil C resulting from tillage and biochar incorporation.

Forest Management

Soil C changes that result from forest management need not be accounted for during the First Commitment period of the Kyoto Protocol. Forest management is included under Article 3.4, which may become mandatory for future Commitment Periods depending on the outcome of international negotiations.

Soil C under forests has been shown to change under silvicultural management, with carbon stocks generally increasing with tree stocking rates (up to 200 stems ha⁻¹), fertiliser application and retention of a weedy cover between rotations. On the other hand, carbon stocks generally decrease with harvesting or site-preparation techniques that physically disturb the soil, or with complete removal of harvest residues and forest floor materials.

However, available data are insufficient to quantify the effects under New Zealand's conditions. Most available studies have not been conducted with carbon accounting as their main objective, and soil carbon measurements have been collected either for shallow depths, without bulk-density measurements, or under very specific soil and climatic conditions that make national extrapolations difficult. It is also not well known to what spatial extent different silvicultural practices are used throughout New Zealand.

In terms of post-2012 accounting, it is possible that the accounting of forest management will become mandatory under either Article 3.4 or under any form of net-net accounting. However, until further work is undertaken to better establish the full effects of forest management activities on soil carbon stocks, taking into account the areas of land affected at the national level, it is very difficult to be certain about the implications that effects of practices such as spot-mounding and ground-based harvesting might have on New Zealand's net position. Mitigation opportunities, such as retention of a weedy ground cover between rotations, are also limited due to their potential to interfere with normal site management for optimum wood production. Nevertheless, there are some practicable and well-established forest management practices available to forest managers that may help maintain or even increase soil carbon stocks (e.g., full residue retention on-site).

There is some potential to include biochar in future operations, especially if small, mobile units can be developed that can make use of available harvest residue for combined bio-energy/biochar production. However, much more work needs to be done to assess whether the use of harvest residues for the production and application of biochar is indeed the most beneficial strategy in greenhouse gas terms, and whether biochar application has useful co-benefits or, instead, lead to some detrimental side effects.

Pastoral Agriculture

Pastoral agriculture is New Zealand's dominant land-based activity, but carbon-stock changes do not need to be accounted for during the Kyoto First Commitment Period because New Zealand chose not to include grazing-land management, which is an optional component under Article 3.4 for the First Commitment Period. It has also long been assumed that soil carbon stocks would be highest under pastoral land use and that they would remain constant in the absence of any land-use change. Recent work, however, has suggested that there may be changes within the broad classification of pastoral agriculture.

In particular, for dairy pastures on lowland non-allophanic soils, soil carbon stocks appear to have declined over the last 20 years. In contrast, on dry stock, hill country pastures, it appears that soil carbon stocks have increased over the last 20 years. So far, no mechanism has been identified for these changes, and there is consequently little confidence in the magnitude of past trends, or for the prediction of future trends. It is also not clear why there appear to be different trends in different landscapes or production systems. If there are indeed differences, it is thus not clear whether those differences are related to terrain, soil type or production system, such as management intensity, fertiliser applications rates, and the degree of soil stability or disturbance.

It is also not yet known whether these losses and gains have occurred recently, have been occurring gradually over time and are ongoing, and whether they occur uniformly across New Zealand. There is also little understanding of the processes controlling, and factors contributing to, such losses. At this stage, it still seems likely that the pastoral sector as a whole is neither gaining nor losing carbon, but further work is clearly warranted to better establish whether different parts of the country may, in fact, display divergent trends, or more generally to better understand the factors that can lead to carbon gains or losses in pastoral soils. There is also a dearth of long-term monitoring of soil carbon below 7.5 cm depth. Currently available information is therefore of limited use both for determining management effects on total soil C stocks, and for meeting international requirements for estimating carbon stocks to a minimum depth of 0–30 cm.

Changes in soil C under pastoral soils may have to be accounted in post-2012 agreements under either Article 3.4 or under any form of net-net accounting. That is of little consequence if there is, indeed, no change in soil carbon. If there are identified changes, however, they could easily become important for national totals because of the large tracts of land involved.

There is little identified mitigation potential as standard pastoral management already leads to high soil C levels. There are risks, however, due to the emerging trend of soil cultivation as part of forage cropping to support pastoral agricultural systems. Cut-and-carry systems, which have recently begun to be considered for high-end production systems, have also not yet been assessed in terms of their C-stock implications. High-country tussock grasslands are also more vulnerable than lower-elevation, more productive pastoral lands, and reductions in soil carbon have been observed in association with various forms of degradation of these lands, especially in relation to frequent burning and nutrient losses.

Cropping

Cropping on land that has remained under cropping since 1990 does not need to be accounted under the Kyoto Protocol because New Zealand chose not to include cropland management, which is an optional component for the First Commitment Period. It is also assumed that carbon stocks on these lands have stabilised at a new level by now. Cropland area is increasing by about 500 ha per year, and upon conversion, it is likely to lose about $0.5 \text{ t C ha}^{-1} \text{ yr}^{-1}$, for about 20 years. Provided conversion rates continue at these relatively small amounts, carbon losses can be expected to be about $0.02 \text{ Mt CO}_2 \text{ yr}^{-1}$, and thus make only a small contribution to the national total.

Mitigation opportunity consists of a reduction in soil disturbance (zero or minimum tillage), which reduces the rate at which organic matter in the soil breaks down, and maximum residue input and incorporation, which principally involves a cessation of residue burning and retaining it on site. The mitigation potential of these options has not yet been satisfactorily quantified, mainly because most past work had not been conducted with the aim of carbon accounting and data have been collected at too shallow a depth, or without bulk-density measurements. A shift towards carbon mitigation practices might also entail other management difficulties that render theoretical options unsuitable in practice.

Horticulture

Like cropping, horticulture is gradually expanding, but by only about 1000–2000 ha per year. On average, horticultural soils are estimated to lose $9 \pm 7 \text{ t C ha}^{-1}$ on conversion from pastoral land, but that loss must be balanced by likely increases in biomass of a comparable magnitude. The combined carbon change is therefore likely to be very small and possibly even positive.

Horticultural operations generally do not aim to maximise biological productivity and often keep bare soil underneath their economic plants, both of which reduce the potential for organic C build-up. A shift to organic farming methods, or any practice that increases the input of residue carbon to the soil, is likely to increase the amount of soil C. There may also be opportunities for biochar incorporation, but the potential of any of these options has not yet been satisfactorily quantified, and questions remain as to their compatibility with standard management operations.

Whether C stock changes to or from horticulture need to be accounted depend on the type of horticultural crop being grown and whether it meets the definition of a forest (and depending on the previous land cover). The expansion of horticulture comes largely at the expense of pastoral land, and since combined carbon-stock changes in those conversions are likely to be small it does not matter greatly whether horticultural expansion is included in the accounting or not.

Erosion

Erosion raises complex questions both in terms of the overall carbon cycle and in terms of carbon accounting. In terms of the carbon cycle, the question is essentially whether erosion acts as an overall source or sink in terms of carbon fluxes to and from the atmosphere. In terms of carbon accounting, the question is how this atmospheric impact can be captured in

accounting rules, or to what extent current accounting rules are inconsistent with the wider role of erosion as a component of the global carbon cycle.

Erosion in the first instance is simply the movement of carbon from one part of the landscape to another, with no immediate exchange with the atmosphere. In the longer term, net carbon emissions to the atmosphere can be increased if carbon input into the system is reduced through reduced biological activity on the erosion scars or if the eroded carbon is rendered more decomposable through the movement from its original location. Net carbon emissions can decrease if the displaced carbon becomes more resistant to degradation, which may occur if it is deposited in anaerobic deposits like lakes or ocean sediments. Decomposition may also be slowed if it is simply buried by other soil material, but the extent of this process is less certain. Overall, erosion and deposition become a carbon sink if decomposition of carbon in depositional sites is relatively slow, and if soil carbon is replaced on the eroded site by inputs to the soil carbon pool from vigorous plant production that returns relatively rapidly to pre-erosion rates.

We report here a first quantification of the likely effect of erosion on carbon fluxes from New Zealand's soils. This has included a detailed analysis of the types of erosion, their sediment yield, the associated carbon concentrations and an assessment of the likely places where eroded carbon might be deposited. It also included assumptions about the rate of recovery of soil carbon on erosion scars, the rate of carbon loss from deposits, and the proportion of material oxidised after deposition in the oceans.

The analysis suggests that overall, erosion in New Zealand constitutes a net sink of 0.85 Mt C/yr for the North Island, and a net sink of 2.3 Mt C/yr for the South Island, for a total of 3.15 Mt C/yr for New Zealand. For the South Island, the analysis indicates a river transport to the oceans of about 2.9 Mt C/yr, with about 0.6 Mt C/yr being oxidised in the ocean and new sequestration on land of 2.9 Mt C/yr, for a net flux out of the atmosphere of 2.3 Mt C/yr. For the North Island, the analysis indicates a river transport of about 1.9 Mt C/yr, sequestration of 1.25 Mt C/yr, and oxidation in the ocean of about 0.4 Mt C/yr. This adds to a net flux from the atmosphere of 0.85 Mt C/yr, but also a loss of the amount of carbon stored on land by about 0.65 Mt C/yr. It is possible to have a net flux out of the atmosphere while land stocks are decreasing through an increase in the amount of carbon stored in ocean deposits.

While this analysis has attempted to quantify each of the relevant terms as carefully as possible, it must nonetheless be recognised that there is a dearth of data on some of the key parameters, such as the rate of recovery of soil carbon stocks on erosion scars and the rate of decomposition of eroded carbon both where it is deposited on land and in water ways. Because of the quantitative importance of this process, it would be warranted to expend further resources on a better quantification of the key rates and processes.

In terms of carbon accounting, erosion is generally counted as a source of soil carbon to the atmosphere. Accounting considers carbon stocks per unit of land, and if carbon is removed from a unit of land, it is considered as a soil carbon loss even if the carbon is simply transferred to a different pool, such as ocean deposits, without actually being lost to the atmosphere. Some difficulty will be encountered in attempts to modify the current accounting philosophy to adequately capture the net impact of erosion and deposition on the atmosphere, as well as on each unit that exists within the current accounting system. Therefore, the inclusion of projections for the erosion and deposition of carbon in national accounts is not presently recommended. It would nonetheless be warranted to develop accounting procedures

for erosion and deposition in order to capture and account properly for the effects of this important process in an unbiased manner. Correcting carbon accounting frameworks to include erosion and deposition processes fully will take some years to develop, but may have net benefits to New Zealand.

1. Introduction

1.1 Scope of Study

At present, the only changes in carbon stocks that are accounted by New Zealand under the Kyoto Protocol are those occurring as a result of the Article 3.3 activities of afforestation/reforestation. These activities typically result in small losses of soil carbon. New Zealand did not elect Article 3.4 of the Protocol, and thus need not report on changes (if any) in the soil carbon pool that occur on managed land that remains under the same land use. However, New Zealand is presently poorly placed to account for any such changes—or, alternatively, to provide evidence that no changes are occurring—as it remains incompletely known what soil carbon data are currently available, where and by whom the data have been collected, who currently holds and maintains the data, and whether they are sufficiently representative to be used for accounting purposes.

The scope of the present study was therefore to review, summarise and document the available New Zealand studies on the effects of land use and land-use change on New Zealand soil carbon stocks and rates of change. These were then to be used, to the extent possible, to develop current best-available soil carbon data with which to determine the effects on New Zealand's net position of likely post-2012 LULUCF activities, mitigation opportunities and accounting options.

Specifically this chapter:

- identifies and catalogues the key characteristics, strengths, limitations and gaps in existing soil carbon datasets available to support future development of soil carbon inventory for post-2012 LULUCF activities, mitigation opportunities and accounting options;
- uses existing analyses of these datasets to generate current best-estimates of soil carbon stocks and change in forest land, grassland and cropland from 1990 to the present, due to changes in land use and management (including intensification);
- extends these estimates to at least 2020 to the extent possible, under a range of likely future land conversion rates;
- identifies mitigation options based on changes in management practice, possible future constraints on nutrient and irrigation application rates, and the likely effects of climate change;
- quantifies the net effect of erosion and soil recovery on erosion scars on national soil carbon stocks, including identification of likely anthropogenic and non-anthropogenic components, and identifies any mitigation opportunities;
- provides where possible statistical assessment of uncertainty, and estimation of likely ranges of data where formal assessment of statistical uncertainty is prevented by critical knowledge gaps.

The above topics are addressed in 5 major sections—the effects on soil carbon of:

- afforestation/reforestation/deforestation
- pastoral agriculture
- arable and forage cropping
- horticulture
- erosion

1.2 Reporting and Accounting Requirements

In future commitment periods, it might become necessary to account for emissions or removals of CO₂ from soils as a result of land management change, or due to land-use change other than afforestation/reforestation. Whether, or to what extent, this occurs depends on the accounting rules agreed as part of the international negotiations about the LULUCF sector. These negotiations are not yet at an advanced stage, and at this stage options that would involve a wider accounting of soil carbon emissions and removals are still among the possibilities.

Because there is a very large land area (>10 Mha) over which soil-carbon changes may have to be accounted in the future as a result of changes in land use or management (other than afforestation/reforestation), changes in carbon stocks of even just 1 t C ha⁻¹ yr⁻¹ could have large implications for New Zealand's carbon balance: such emissions would be 8 times those New Zealand is required to offset in each year of the first commitment period to meet international obligations (MfE 2008). Conversely, New Zealand would benefit considerable if soils were an overall carbon sink.

For practical reasons, due to the typically very large variability in soil carbon stocks, it is usually not possible to detect changes of less than 1 t C ha⁻¹ yr⁻¹. Recent work suggests that at least 100 samples would be required to detect changes of even 20% (c. 20 t C ha⁻¹) in carbon stocks with sufficient statistical confidence. Thus, by the time changes in soil carbon stocks become evident, either large gains or large losses would have occurred. Although such gains or losses may not necessarily be accountable under post-2012 accounting regimes—depending on the outcome of international negotiations—they will (if known) have to be reported under the UNFCCC in New Zealand's annual greenhouse gas inventory.

2. Effects of Afforestation/Reforestation/Deforestation on Soil Carbon

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2.1 Introduction

2.1.1 Soil carbon change: stock-change factors and paired sites

In New Zealand, afforestation and reforestation of grazing land give rise to large increases in biomass carbon sinks. However New Zealand, and international, studies show that changing land use from grassland to woody species may also lead to small but significant losses of mineral soil carbon—at least for the first 10–15 years after land-use change (e.g., Tate et al. 2003). Such changes must be reported in accounting under Article 3.3 of the Kyoto Protocol.

Characterising changes in soil carbon stocks is complex, because spatial variation in soil carbon tends to be very large. Many measurements are therefore required to determine average values precisely, and detecting change—and especially small amounts of change—is therefore often very costly. For this reason, approaches taken to determining changes in soil carbon stocks have to date focussed on making maximum use of data already available, particularly use of steady-state values of soil carbon stocks derived from New Zealand’s national soils database (NSD). The NSD values are assumed to represent steady-state soil carbon stocks because site selection for sampling has been very careful to choose undisturbed sites.

The NSD data have been used to derive a national-scale predictive equation for steady-state soil carbon stocks as a function of soil type, land use and climate (e.g., Tate et al. 2003). Land use implicitly includes both vegetation cover and management factors. The difference in steady-state soil carbon stocks that results from a change in only the land use is termed a stock-change factor. Stock-change factors only provide a measure of the difference in steady-state soil carbon stocks: they do not provide any measure of the time required to transition from one steady-state value to another. The IPCC default value for the transition time is 20 years, but both longer and shorter transition times are possible (e.g., Scott et al. 2006b; Kirschbaum et al. 2008b). The IPCC default also assumes, for simplicity, that the change is linear over time even though biological systems do not generally behave in such a way.

In part as an attempt to provide additional validation of the stock-change factor derived from NSD data, a number of “paired-site” studies have also been completed in New Zealand (e.g., as summarised in Baisden et al. 2006). Paired sites comprise a pair of sites that are matched as closely as possible in terms of soil type, soil profile characteristics, slope, aspect, altitude, and climate and differ only in land use. For afforestation/reforestation, a stock-change factor is calculated as the difference between the average soil carbon stocks between the matched grassland and forest sites. Paired-site studies have several advantages over use of NSD data when deriving stock-change factors:

- As these sites have been purposefully selected such that climate, soil type and profile characteristics, and topography are as similar as possible, any difference in soil carbon stocks is likely to be a function only of the difference in land use.
- Paired sites are purposefully selected to ensure an absence of mass-movement erosion. By contrast, data in the NSD may be biased because the grassland data was deliberately not collected from eroded lands, whereas samples from some forests may have been collected from erosion-affected sites—although data in the NSD were screened as far as possible to avoid this.

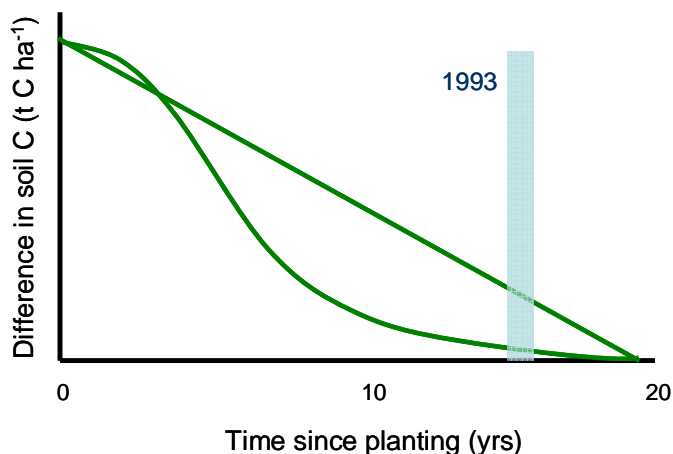
Selection of paired sites is nonetheless not without its problems. The existence of truly paired sites in close spatial proximity is rare, as forest and grassland occurrence is itself often dictated by soil type, slope, rainfall or erosion history. Further variation may also be introduced because these factors may also have caused, or been affected by, differing land-use and vegetation-cover history. Because locating paired sites is very time-consuming and expensive, presently available datasets also comprise a quite limited number of studies, and thus estimates of stock-change factors are not yet very precise. Caution is thus required when interpreting stock-change factors from paired site studies—just as much as when interpreting results from factors derived from the NSD. We illustrate this point further below, when comparing stock-change factors derived from NSD and paired-site studies.

2.1.2 Rates of change in soil carbon stocks

For improving accounting of change in soil carbon stocks with afforestation/reforestation, it is not just improvements in the accuracy of the stock-change factor that is required: improved knowledge of the time-dependence of change is also required. As noted above, the IPCC default period over which soil carbon stocks are assumed to occur as a result of a change in land use is 20 years, with the change also assumed to occur at a constant rate. Because New Zealand experienced a temporary but large increase in the rate of post-1990 exotic forest plantings during the early 1990s, it is important to know whether a constant rate of change is realistic. In particular, if the change between soil carbon stocks in grasslands and forests were not constant over time but rather, as often observed in adjustments of biological systems, exponential in shape, then much change in soil carbon stocks associated with early 1990s plantings would have occurred prior to the start of the first commitment period—and thus need not be accounted.

Figure 1 illustrates the effect that different assumptions about the shape of the temporal trajectory of change in soil carbon following establishment of forest on grasslands have on liabilities during the first commitment period (CP1), 2008–2012. If an exponential rather than a constant rate of change in soil carbon stocks were proven appropriate, it could potentially reduce New Zealand's liabilities by between \$50–\$100M during CP1 (at the present stock change factor of 18 t C ha⁻¹). Unfortunately, New Zealand has at present very little time-sequential data on rates of change in soil carbon with land-use change. Therefore only modelling studies can be used to investigate the temporal behaviour of carbon stock changes at present, and as discussed later (Section 2.2.1.4) these are not yet definitive—again, primarily because of a lack of adequate data for model calibration and validation in this case.

Figure 1 Example of the implications of different possible temporal trajectories for the change in soil carbon stocks following afforestation/reforestation. A constant (straight line) and exponential reduction in soil carbon stocks over time is shown. A forest planted in 1993 at the peak of early-1990s planting rates is 16 years old in the middle of CP1. The annual rate of change in soil carbon stocks through CP1 for such a forest (the slope of the line, or curve) is much smaller for an exponential decay in soil carbon stocks than for straight-line decay



2.2 Afforestation/Reforestation

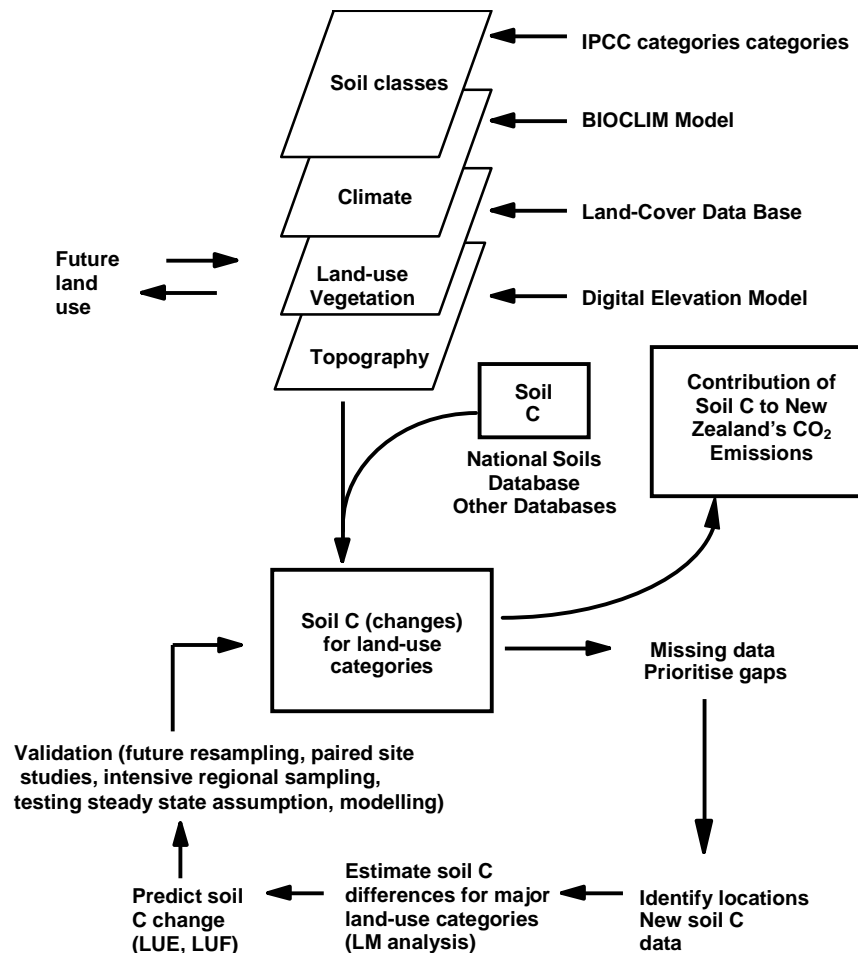
2.2.1 Stock change factors from national soils datasets: a review

To determine carbon stocks and change with land-use change, New Zealand has developed a soil Carbon Monitoring System (CMS) that is an extended version of the methodology recommended by the Intergovernmental Panel on Climate Change (IPCC). The Soil CMS has recently been adopted for international reporting under the Kyoto Protocol as part of the Ministry for the Environment's LUCAS inventory system, and is presently passing through a series of quality assurance steps to make it more transparent under expected international review.

A schematic of the Soil CMS system is depicted in Figure 2. Essentially, it comprises a regression equation derived using a Linear Modelling approach that predicts soil carbon stocks as a function of land use, soil type, climate class, and a slope-rainfall product (Tate et al. 2003, 2005; Baisden et al. 2006). The equation was derived from NSD data, and from datasets held by SCION. The regression equation can be used to provide estimates of steady-state carbon stocks for particular land uses, with the differences in carbon stocks between grassland and other land uses being the stock-change factors referred to earlier.

The Soil CMS has been updated a number of times as knowledge has improved of the processes and factors driving soil carbon changes. For example, the recent inclusion of a slope-rainfall product variable is expected to correct soil carbon stocks for the effects of surface soil erosion (Tate et al. 2005). Most recently, the calibration datasets have been further expanded by incorporating data gathered during pilot-scale inventory programmes conducted in Marlborough, and by a further round of quality control to remove soil profiles that might have been affected by mass-movement erosion (Baisden et al. 2006).

Figure 2. Schematic of the Soil CMS. The system calculates soil carbon stocks from a generalised regression equation, based on variables of soil type, climate, land-use, and a slope/rainfall product. Calibration data are taken from soil profile data in the NSD and other datasets. Soil carbon stocks are estimated for New Zealand using spatial datasets of the variables in the regression equation



National soil carbon stocks, and stock-change factors, derived using the most recent version of the Soil CMS, are given in Table 1 (Tate et al. 2005; Baisden et al. 2006). The best estimate of the stock change factor for afforestation/reforestation of grasslands currently available is $18.4 \pm 5.7 \text{ t C ha}^{-1}$, or $0.9 \text{ t C ha}^{-1} \text{ yr}^{-1}$ over a default 20-year period. This figure compares with a national average carbon gain of $8 \text{ t C ha}^{-1} \text{ yr}^{-1}$ over the same time period for a *Pinus radiata* forest. Changes in soil carbon stocks are thus a small but significant fraction of the total change in carbon stocks that occurs with afforestation/reforestation.

Table 1 National average soil carbon stocks under pasture, and stock-change factors, derived using the most recent version of the Soil CMS. All stock-change factors are expressed relative to pasture soils

Land use	Mean carbon stock (t C ha ⁻¹)	Stock-change factor (t C ha ⁻¹)	Standard error in stock change factor (t C ha ⁻¹)
Pasture	108.0	-	-
Cropland	98.5	-9.5	7.1
Horticulture	102.1	-5.9	7.9
Shrubland	101.1	-6.9	4.7
Exotic forest	89.6	-18.4	5.7
Indigenous forest	111.5	3.5	4.9

The Soil CMS relies on a number of assumptions (Tate et al. 2003, 2005) and therefore requires validation using independent data. Two regional-scale validation programmes have been completed to date. The first comprised random sampling of the largest climate/soil/land-use category in New Zealand: temperate volcanic high clay content soils under pasture. The exercise confirmed that for this category of major importance to New Zealand, modelled and measured estimates of soil carbon stocks were well within the limits of statistical agreement (Tate et al. 2003, 2005). A second validation exercise was carried out using data available for the 0–0.1 m layer at 204 sites over 6000 ha in North Canterbury (Tate et al. 2005). Regional mean carbon stocks were not significantly different from the CMS prediction for shrublands, planted forests and unimproved grazing land, but were significantly lower than predicted for improved grazing land, broadleaf (indigenous) forests and, to a lesser extent, arable land. The very stony nature of soils in some of these classes appeared to account for most of this difference. For example, for the land cover that showed most difference (c. 12 t C ha⁻¹) between predicted and measured values (improved grazing land), the mean stone content was 19 t stones ha⁻¹, whereas the data that had been used to develop the Soil CMS predictive equation had only 1.9 t stones ha⁻¹.

2.2.2 Stock change factors from paired site studies: a review

Paired site studies in New Zealand have recently been summarised and critically reviewed (Baisden et al. 2006) to provide a better basis for comparison with and further validation of Soil CMS-predicted stock-change factors. The review has raised some questions about whether both land-use and erosion history are sufficiently well-known at about 4 (out of 12) *P. radiata* forest sites to allow their inclusion—although similar land-use history issues are expected to be reflected in samples in the combined NSD/SCION datasets used by the Soil CMS, and the single site thought to possibly be affected by mass-movement erosion (because of its relatively steep slope) did not appear to have a disturbed soil profile. For the moment, there seems insufficient reason to exclude these 4 paired sites, even though they tend to result in larger soil carbon stock-change factors for the paired site dataset—but not overly so by international standards (see Section 2.3).

Mean values from the paired-site dataset suggest considerably smaller soil carbon losses occur during exotic forestation than predicted by the Soil CMS (Baisden et al. 2006). However, the exotic forest data in the Soil CMS dataset are almost exclusively *Pinus radiata*, whereas estimates from the paired-site studies include those for non-*radiata* species which

apparently show different behaviour. It is therefore more appropriate to split the paired-site datasets into *radiata* and non-*radiata* species to facilitate comparisons with Soil CMS-predicted stock-change factors, as discussed further below.

2.2.3 Forestation with *Pinus radiata*

When paired sites with only *Pinus radiata* forests (aged 10 years and over) are included in the analysis, the mean estimate for soil carbon losses during forestation is 8.5 t C ha⁻¹, but could range from a 16 t C ha⁻¹ loss to a 1.3 t C ha⁻¹ gain (95% CI). Data for paired sites in this category are summarised in Table 2 (Baisden et al. 2006).

Table 2 Estimates of soil carbon stock change factors (t C ha⁻¹) from paired site studies, due to forestation of pasture (Baisden et al. 2006)—*P. radiata* sites only

Soil Depth Increment	Estimate (and standard error)	95% Confidence Interval	Number of pairs
0–0.1 m	–3.9 (2.2)	–8.3 to +0.5	22
>0.1 m	–4.6 (2.5)	–10.2 to +0.9	9
0 – ~0.3 m*	–8.5 (3.4)	–15.6 to +1.3	-

*0–~0.3 m is the sum of the two depth increments

2.2.4 Forestation with species other than *P. radiata*

The data compiled for other species suggested a slight gain in soil carbon stocks following forestation of pastures (Table 3; Baisden et al. 2006). For the 0.1– ~0.3 m depth, the difference between *P. radiata* and other species is significant ($p = 0.04$). However, for the combined 0– ~0.3 m depth increment, evidence for a difference between *P. radiata* and other species is somewhat weaker ($p=0.09$). This suggests there may be potential value in reporting carbon stock changes separately for non-*radiata* species if the land areas on which they are planted can be determined, and if further data collection supports the initial evidence that soil carbon may respond differently to forestation under *P. radiata* than for other species. However, it must first be established whether the non-*radiata* paired-sites are representative of current plantings of non-*radiata* species.

Table 3 Estimates of soil carbon stock change factors (t C ha⁻¹) from paired sites due to forestation of pasture (Baisden et al. 2006)—non-*radiata* sites only

Soil Depth Increment	Estimate (and standard error)	95% Confidence Interval	Number of pairs
0–0.1 m	–3.6 (2.6)	–9.4 to +2.2	12
>0.1 m	+5.0 (3.1)	+12.3 to –2.3	8
0– ~0.3 m*	+1.6 (4.5)	–7.8 to +11.8	-

*0– ~0.3 m is the sum of the two depth increments

2.2.5 Forestation with all species

Due to the limited size of the paired-site dataset, it might be argued that the best estimate for soil carbon changes following forestation is obtained by including all available data for forests ≥ 10 years in age. This yields an estimate similar to the *P. radiata* paired site and Soil CMS values for the 0–0.1 m depth increment, but a loss of lesser magnitude for the below 0.1 m soil depth (Table 4). As expected from the data below, it appears the 0–~0.3 m depth increment is sensitive to either the species planted, or the factors related to the “outlier” data (see below). This estimate is not, however, directly comparable to the estimate of the soil carbon stock change factor from the Soil CMS for depths below 0.1 m, because of the inclusion of a large proportion of non-*radiata* species in the total paired-plot dataset.

Table 4 Estimates of equilibrium soil carbon changes (t C ha^{-1}) from paired site studies due to forestation of pasture (Baisden et al. 2006)—data from all sites

Soil Increment	Depth	Estimate (and standard error)	95% Confidence Interval	Number of pairs
0–0.1 m		–3.8 (1.7)	–7.2 to –0.4	34
>0.1 m		–0.7 (2.4)	–4.5 to +5.3	17
0–~0.3 m*		–4.6 (3.3)	–11.6 to 2.4	-

*0 – ~0.3 m is the sum of the two depth increments

2.2.6 Reconciling differences in stock-change factors from Soil CMS and paired site studies

There are considerable differences in the mean values of the soil carbon stock change factors currently obtained from the Soil CMS and paired-site studies: -18.4 and -8.5 t C ha^{-1} , respectively. However, both estimates are subject to substantial uncertainty because of the relatively small size of the datasets involved and the natural spatial heterogeneity of soil carbon stocks. If uncertainties as expressed by the standard error values recorded in Tables 3 and 4 are considered, rather than just mean values, the difference between paired-site and Soil CMS estimates of the mean soil carbon stock change factor with afforestation/reforestation of pastures is, in fact, not statistically significant ($p > 0.1$).

Agreement of estimates of the mean stock change factor for the two approaches is high for the 0–0.1 m soil depth increment. By contrast, a substantial difference between the mean stock-change factors exists for the overall 0–0.3 m depth increment due largely to significant differences in the 0.1–0.3 m increment. It is also notable that when data from all New Zealand sites and forest species are considered, *P. radiata* plantations have a larger mean reduction of 8.5 t C ha^{-1} (95% CI: 15.6 t C ha^{-1} loss to 1.3 t C ha^{-1} gain), whereas other forest species show a mean gain of 1.6 t C ha^{-1} (95% CI: 7.8 t C ha^{-1} loss to 11.8 t C ha^{-1} gain). It is possible that planting of *radiata* and non-*radiata* species onto pasture have different effects on the change in soil carbon stocks, but this must remain a tentative conclusion given the size of current datasets—although one international review has also suggested larger soil carbon losses may occur with planting of conifers than for planting of other exotic forest species (Guo & Gifford 2002).

An alternative explanation for differences between *radiata* and non-*radiata* studies may also be that the New Zealand studies on non-*radiata* species have been in drier regions, and previous work has indicated that C losses increase with increasing rainfall (Guo & Gifford 2002; Kirschbaum et al. 2008b). A recent New Zealand study conducted after the analysis by Baisden et al. (2006) at relatively dry (c. 600 annual rainfall) non-*radiata* coniferous sites did not find any statistically significant soil carbon losses with afforestation/reforestation (Davis et al. 2007). There is increasing international evidence (reviewed below in Section 2.3.1) that changes in soil carbon stocks with coniferous afforestation/reforestation of pastures may only become significant as annual rainfall increases above about 1000 mm—though losses may increase quite rapidly above that (e.g., Kirschbaum et al. 2008b).

2.2.7 Rates of change in soil carbon following afforestation/reforestation

As noted in the introduction to this section, whether the losses in soil carbon stocks that occur as a result of forestation of pastures follow a relatively linear or more concave-downwards exponential decay can make a considerable difference to the change in soil carbon stocks that must be reported by New Zealand under the Kyoto Protocol. This is because there was a large peak in planting rates during the early 1990s, with those forests being about 16 years old mid-way through CP1. If the soil carbon change more closely follows an exponential time dependence following forestation, the rate of change in carbon stocks during CP1 will be considerably less than if it followed a linear decay (see Fig. 1).

Unfortunately, very few measured data are available either nationally, or internationally, to support preferential selection of an exponential over a quasi-linear linear decay. Ideally, one would like to use datasets of frequently measured time series of soil carbon stocks, but such definitive datasets do not exist—and their collection would require a large investment given the relatively small expected changes in carbon stocks and the large spatial heterogeneity in background soil carbon stocks.

The only known New Zealand study in which time sequential data on soil carbon stocks were periodically measured following forest establishment on pasture is that of Beets et al. (2002), though data are only available to a depth of 0.1 m. In that study, soil carbon stocks were measured periodically at 30 permanent sample plots within 3 catchments at Puruki in the central North Island. The forest was a first rotation *P. radiata*. Measurements over a stand ages of 5 to 22 years showed that decreases in mineral soil carbon stocks were detectable by 5 years, when stocks had decreased by about 4 t C ha⁻¹ to 0.1 m depth but did not decline further after that. It was also concluded (Oliver et al. 2004) that no further net change in mineral soil carbon in the total soil profile occurred in the 3 catchments when the sample plots were re-measured after 5 years, though some mixing of the soil profile was apparently present.

At a later paired site study using one of the forested catchments that had been periodically monitored (Rua), when the trees were more than 20 years old, mineral soil carbon stocks under forest were on average 7 t C ha⁻¹ lower to 0.3 m depth than under pasture transects. The means were 12 t C/ha lower to 0.1 m depth under forest than under pasture, a much larger decrease than found in Beets et al. (2002), but 5 t C ha⁻¹ higher from 0.1 to 0.5 m depth. Other studies at Puruki have noted large differences between pasture and forested sites at depths below 0.2 m (c. 30 t C ha⁻¹; Ross et al. 1999; Yeates et al. 2000). This continues to emphasise the highly spatially heterogeneous nature of soil carbon stocks, and

the need for a much more comprehensive approach to sampling before definitive conclusions can be drawn on the exact magnitude of change with afforestation/reforestation.

A further study has recently been completed in which the temporal dynamics of soil carbon change were modelled, using the well-respected process-based model Roth-C (Scott et al. 2006b). The model was calibrated against measurements of soil carbon stocks in a 26-year-old *P. radiata* forest at Tikitere in the central North Island, using data on the timing, magnitude and quality of litter carbon inputs to the soil pool. The change in soil carbon stocks showed a broadly downwards concave shape, although it could also be interpreted as a two-phase linear decay process. The calibrated model was also used to simulate changes in soil carbon stocks for the Puruki catchment, unadjusted except to account for changes in climate and the higher litter inputs from higher stocking than the Tikitere catchment. This simulation showed considerable soil carbon losses (c. 25 t C ha⁻¹) after 20 years, with a relatively well-defined exponential decay. Losses of this magnitude are considerably more than those recorded at Tikitere by Beets et al. (2002) and Oliver et al. (2004), but somewhat less than those recorded by Ross et al. (1999) and Yeates et al. (2000). Overall, the model-predicted time-dependence of change in soil carbon stocks under New Zealand conditions is consistent with an exponential decay, though given the very limited model calibration/validation data such a conclusion can only be very tentative.

2.3 Deforestation

2.3.1 Initial studies

Studies of the effects of deforestation on soil carbon stocks have only very recently begun in New Zealand, and then only by an individual researcher as part of a PhD programme. The work completed to date comprises studies at three farms located near Atiamuri, Manawahe and Tokoroa in the Taupo-Rotorua Volcanic Zone, where a plantation forest has recently been converted to pastoral farming. At each farm, a permanent pasture site was sampled as well as one or two conversion sites (1-yr, 3-yr, 5-yr) on comparable soil types. The conversion sites had previously been forested for 23 years (*P. radiata*) at Atiamuri; 26 years (*P. radiata*) and 10 years (*Eucalyptus nitens*) at Manawahe; and 63 years (*P. radiata*) at Tokoroa. Pumice soils at Atiamuri are mapped as Taupo sandy silts; Tephric Recent soils occur at Manawahe; and at Tokoroa the soils are older and more weathered intergrade Allophanic soils, with some pumice present in the profile.

At each site, three transects were chosen, and along each transect, five positions were sampled to 15-cm depth (five cores bulked per position, for two depths 0–7.5 cm and 7.5–15 cm). Soil samples were then analysed for total C and N, as well as for routine soil fertility tests. In addition, pasture was sampled monthly from a fixed area under exclusion cages (5 replicates per site) to estimate pasture production. At the Atiamuri and Tokoroa farms, capital dressings of diammonium phosphate (137 kg/ha P) were added to the new clover-ryegrass pastures with additions of Mg, trace elements and lime. After the initial year, conversion pastures at all three properties typically received two N dressings annually (autumn and spring) of between 74 and 88 kg N/ha/yr. These capital fertiliser inputs and subsequent maintenance P and S fertiliser additions successfully raised the soil P status to optimum agronomic values within 3–5 years after conversion, at two of the three farms (Hedley et al. 2007). Pastures respond rapidly to management inputs and pasture production reached 82–95% of established pastures within the first 2 years after conversion.

2.3.2 Initial results

The results indicate that total soil C and N accumulate rapidly in these soils under new pastures (Table 5). A narrowing of C:N ratio from 17.5 to 15.7 and 15.7 to 14.5 over the first 5 years after conversion at Atiamuri and Manawahe, respectively, reflects the proportionally greater accumulation of N compared with C in these soils, as the clover-ryegrass sward establishes with accompanying N fixation, as well as N inputs from dung, urine and fertiliser. Total soil C increases by approximately 4 mg cm⁻³ per year for the first 5 years after conversion, assuming 5 years of pasture growth between sampling dates for the 1-yr and 5-yr conversion sites at Atiamuri and Manawahe. Similarly, soil N increased by 0.31 mg cm⁻³ per year for the first 5 years at these 2 sites.

This equates to a soil carbon sequestration rate of 6.2 t C/ha/yr and a soil N sequestration rate of 0.47 t N/ha/yr to 15 cm soil depth. The ability of these newly converted soils to sequester 0.47 t N/ha/yr suggests that a large proportion of the N fertiliser applications and biologically fixed N are immobilised into soil organic matter. The apparent ability to sequester 6.2 t C/ha/yr for the first five years after conversion partially compensates for the lost forest sink capacity. Reported growth rates of *P. radiata* in this region are between 22 and 39 m³/ha/yr (Kimberley et al. 2005).

Assuming 25% fresh weight is carbon, then these forests would accumulate between 5.2 and 9.7 t C/ha/yr, so that the recorded soil carbon sequestration rate is a significant offset. Walker (1968) also found similar soil carbon increases when Taupo sandy silts were first developed from scrub to pasture in the late 1950s. Scott et al. (2006b) also estimated greater inputs of carbon under steady-state pasture (9 t C/ha/yr to 50 cm soil depth) compared with established (>12 years) forest (1.53 t C/ha/yr) using the Roth-C model.

Table 5 Total carbon (C), total nitrogen (N) and C:N ratio of soil samples (15 bulked replicates to 15 cm soil depth) from three farms where forest has recently been converted to pasture in the Taupo-Rotorua Volcanic Zone

Location	Land Use	Total C		Total N		C:N ratio
		0–15 cm soil depth				
		mg cm ⁻³	<i>s.e.</i>	mg cm ⁻³	<i>s.e.</i>	
Atiamuri	1-yr conversion	25.0	2.4	1.4	0.1	17.5
	5-yr conversion	40.6	5.1	2.6	0.3	15.7
	Permanent pasture	38.5	2.2	3.5	0.2	10.9
Manawahe	1-yr conversion	20.4	1.7	1.3	0.1	15.7
	5-yr conversion	46.4	2.5	3.2	0.2	14.5
	Permanent pasture	48.0	3.3	4.1	0.3	11.7
Tokoroa	3-yr conversion	45.7	4.1	2.8	0.3	16.2
	Permanent pasture	47.1	2.9	4.0	0.2	11.7

2.3.3 Initial conclusions

Overall, these results indicate that a significantly greater research effort should be made to determine the fate of soil carbon stocks following deforestation, supported by modelling to forecast the long-term soil carbon response. Further research should sample to 30 cm soil depth to comply with Kyoto Protocol accounting procedures. In addition sampling strategies should aim to better account for the variable nature of these disturbed soils, where trees have commonly been ripped out and left in windrows for a few years before burning. A field-reflectance spectroscopy method is being developed for rapid in situ soil carbon and nitrogen analysis to address this issue (Kusumo et al. 2008). If soil carbon stocks can be maintained in the longer term at even a reasonable proportion of the gains observed in the preliminary experimental studies summarised here, the increase offers a useful partial offset of deforestation emissions.

2.4 Review of key international information

2.4.1 Implications for stock change factors

Three key international reviews of change in soil carbon stocks with forestation of pastures have been completed in recent years. The overall conclusion of the first two reviews is that when pastures are converted to either secondary forests or plantations they often lose soil carbon, with the average loss being about 10% (Paul et al. 2002; Davis & Condrón 2002). That is, for New Zealand situations, losses of the order of 10 t C ha⁻¹ would be expected. The third review involved a statistical meta-analysis of internationally-published studies on soil carbon losses with forestation, and indicated the trend was significant for coniferous but not for broadleaved forests (Guo & Gifford 2002).

Trends in soil carbon also changed over time after forest establishment. Stands between 6 and 30 years of age showed both losses and gains, whereas stands more than 30 years old showed increasing soil carbon stocks (Paul et al. 2002). Also evident was that the effect of land-use change depended on the amount of rainfall received (Guo & Gifford 2002). There was generally no significant mean change in soil carbon for sites where annual rainfall was less than 1250 mm, but losses of 10–15% occurred with rainfall of 1200–1500 mm, and losses of 25% where rainfall exceeded 1500 mm. It is notable that if sites in the study of Davis et al. (2007) mentioned earlier, that showed no overall mean change in soil carbon stocks for a relatively dry environment, are ordered by rainfall, then there nonetheless is a trend for change in soil carbon stocks to increase with increasing rainfall (Murray Davis, SCION, pers. comm.).

The trends in soil carbon stock change with rainfall have recently been successfully modelled from first principles by Kirschbaum et al. (2008a, 2008b). Using a coupled soil carbon-nitrogen dynamics model, CenW, it has been possible to demonstrate that observed changes in soil carbon with rainfall are consistent with a reduction in soil C:N ratio, imposed by a combination of forest litter being more lignified than grassland litter, greater removal and storage of nitrogen (in woody biomass) under forest conditions, and increased leaching of soil nitrogen at higher rainfalls (Kirschbaum et al. 2008a, 2008b). The trends in soil carbon loss predicted by the model agree reasonably well with the global average observations by Guo and Gifford (2002), although are possibly too conservative (i.e. the model predicts smaller losses). This may be due to the assumption that 20% of pine litter falling on the forest floor

is incorporated into the soil, which is perhaps too high, as the work of Scott et al. (2006b) suggests 10% incorporation may be more likely.

Although average predicted and measured trends in soil carbon losses with rainfall are similar, the variation in losses at individual sites can still deviate considerably from these, depending on antecedent vegetation, decomposability and transfer of forest and grassland litter to the soil, and forest biomass growth rates and thus nitrogen storage. For example, the modelling studies by Kirschbaum were carried out for a *P. radiata* forestation site with 660 mm annual rainfall, in Canberra, Australia. Although global average losses for such rainfall are negligible (Guo & Gifford 2002), measured soil carbon change at the site was about 12 t C ha⁻¹ about 18 years after planting onto low productivity grassland.

Overall, there is clearly an urgent need for the CenW model to be full parameterised for New Zealand conditions, to get a better understanding of the effect of coupled soil carbon-nitrogen dynamics under soil, nutrient, rainfall and biomass production conditions typical of New Zealand's *P. radiata* forests.

2.4.2 Implications for Rates of change in soil carbon stocks

The two international studies that deal with the temporal dimension of change in soil carbon stocks with afforestation/reforestation are those of Paul et al. (2003) and Kirschbaum et al. (2008a, 2008b). The modelling of Paul et al. (2003) was formulated using data on above-ground biomass production, litter composition and accumulation rates, and litter decomposability at 7 sites, but not calibrated against soil carbon data. The work of Kirschbaum et al. (2008a, 2008b) used similar input parameters but for a single site, and was calibrated using paired site data with measurements of soil carbon stocks under forest 14 and 18 years after planting.

In the Paul et al. (2003) study, change in soil C at the study sites fell broadly into two categories:

- Decreased slightly (by up to 2.6–6.5 t C ha⁻¹) during the first 6–10 years after forestation, followed by a gradual increase. Pre-establishment levels were predicted to be reached after 10–22 years, and after 40 years the amount of soil C was predicted to be up to 20 t C ha⁻¹ more than under the preceding pasture.
- Rapidly decreased (by 15.7–22.8 t C ha⁻¹) during the first 10 years after afforestation, followed by only a slight increase. As a result, even 40 years after forestation, it was predicted a net decrease (8.2–15.0 t C ha⁻¹) remained in soil C.

This difference in behaviour stems largely from differences in site productivity: low productivity sites showed small decreases and long-term soil carbon gain, whereas high-productivity sites showed large losses and more limited long-term gain. All modelled decay curves showed exponential concave downwards behaviour.

Modelled change in soil carbon in the Kirschbaum et al. (2008a, 2008b) study shows relatively complex behaviour, with the site exhibiting a rapid exponential decay of about 10 t C ha⁻¹ over the first 2–3 years followed by a exponential concave upward gain of about half this amount for a similar period. Following this, there is a period of little change over about 5–7 years, and then a slow linear decline back to a net loss of about 10 t C ha⁻¹.

Taken overall, together with the New Zealand study by Scott et al. (2006b), it is difficult to generalise the temporal response of soil carbon to afforestation/reforestation. A wide range of responses seem likely, depending on antecedent land-use, site productivity, litter quantity and quality, and gross inputs from harvest residues. At present it does not seem possible to reject the IPCC default approach of assuming an average linear decay over about 20 years as being broadly appropriate.

2.5 A Re-analysis of the Soil CMS Stock Change Factor

2.5.1 Introduction

The NSD data used for assessing the stock change factors in the Soil CMS were collected over the years for a variety of purposes. Some areas, particularly around research institutes, were greatly over represented, and some areas, particularly high country, were greatly under represented. To overcome this lack of representativeness, the existing soil data were used to estimate the relationships between soil total C and soil type, climate, land use, altitude and slope. These estimates were then applied to the known composition of New Zealand land forms leading to a national estimate of total C. An intermediate step was to summarise climate, soil classes, land use, rainfall and slope into a linear model that described the main sources of difference in soil total C (Section 2.2.1). Soil class and climate were summarised into 18 categories, land use into 6, and slope and rainfall as their product. The model assumed that land use differences in total C were the same in all soil/climate categories: that is, it was assumed that there was a unique land use effect that applied over all soil/climate classes. This greatly simplified both the model and its use.

The key assumption in this analysis was that the net effect of factors not included in the model could be ignored as random noise. That is, that there was no factor omitted from the model that both affects soil carbon and was itself affected by the misrepresentation of land properties in the sample. There is no guaranteed way of avoiding bias from this source other than by collecting new data through a random survey. However, the linear model can be refined by adjusting for tendencies for clustered sites to be alike. The relationship between the correlation of two sites and their distance apart can be estimated and used to give less weight to sites in clusters and more weight to isolated sites. Since lands with properties that are under represented in the sample will tend to have sites that are sparser than land with properties that are over represented, this approach will automatically re-weight the sample data to compensate for lack of representativeness.

The difference of immediate interest is that between pasture and exotic forest. An alternative way of reducing any spatial effect from this estimate is to include in the analysis only those sites that are near to an exotic forest site. This ensures that the pasture sites used are only those close to at least some land appropriate to exotic forestry.

2.5.2 Re-analysis of National-scale Data

Results of the initial spatial autocorrelation analysis

A re-analysis of data was completed using R software (R Development Core Team 2008) together with the spatial autocorrelation function described by Pinheiro et al (2008). A complete spatial correlation analysis requires estimation of the dependence of the correlation on distance, which is beyond the scope of this study. Analyses carried out so far have only

tried a few standard correlation models without taking the further important but time consuming step of checking each form's appropriateness. It is very likely that the strength of the correlation will depend on land use, soil type or other land properties, but analyses used so far have used the same correlation model for all sites.

The results of these analyses are shown in Table 6. They show that allowing for spatial correlation reduces the difference in predicted soil carbon stocks between pasture and exotic forest, and also brings scrub and exotic forest closer together. Although these changes are within the bounds of sampling error the inclusion of spatial effects does improve the model fit, which means that the estimates are likely to be closer to the true values.

Table 6 Estimates of land-use effects according to three models. The standard model is that quoted in Baisden et al. (2006). Values are given in t C ha⁻¹

	Standard linear model		Spatial model		Standard model, sites < 40 km from exotic forest	
	Value	Std.Error	Value	Std.Error	Value	Std.Error
Baseline: Mean for pasture	111.0	4.2	120.6	14.1	109.1	5.1
Cropland LUE	-9.5	7.2	-5.4	6.8	-8.9	7.2
Exotic Forest LUE	-18.4	5.7	-8.0	5.6	-12.9	5.6
Horticulture LUE	-5.9	7.9	-0.5	7.4	5.2	9.3
Ind. Forest LUE	3.4	4.8	0.2	5.0	-1.8	5.9
Scrub LUE	-6.9	4.6	-10.8	4.5	-10.8	5.9

Observations on the initial spatial autocorrelation analysis

The spatial correlation between two sites was assumed to be $(1-n) \exp(-r/d)$, where:

- n is a nugget effect, the correlation of points arbitrarily close together. Its estimated value of 0.49 is very close to that found in studies of soil cores within sampling plots.
- d is a distance beyond which there is no correlation, estimated to be 177 km.
- r is the distance that two sites are apart. If r is 10 km, the correlation is 0.48, at 50 km, it is 0.38, and at 100 km, it is 0.29.

The estimated mean total C has a very large SE, and estimates for other land uses are all highly correlated. This suggests sites for all land uses are clustered together and are therefore influenced by the same spatial effects. This will not affect differences between land uses, which therefore have similar SEs to the standard analysis. However, this effect needs further investigation.

2.5.3 Initial conclusions

Table 7 shows the change of each land use area from 1992 to 2006. The estimate of the consequent change in total soil C depends on which model is used; the table shows the contribution of each change to the total. For example, the standard model estimates that the loss of 127 000 ha of pasture would ultimately lead to a loss of 14.074 Mt of soil carbon. If all this pasture had been planted in forest, the compensating gain would be 11.757 Mt C. The difference between these, 2.317 Mt C, is the net carbon loss. The equivalent calculation using the spatial correlation model estimates a net loss of 0.988 Mt C, and using only sites near exotic forests results in an estimated net loss of 1.2215 Mt C.

Table 7 Estimates of total C change (Mt C) in soil 1996 to 2002 from three different models. The small overall loss is mainly to expansion of urban land

Land use	Area change (ha)	Total C loss (t): Standard model	Total C loss (t): Spatial correlation model	Total C loss (t): Standard model, sites < 40km from exotic forest
Pasture	-127 000	-14.074	-15.293	-13.828
Cropland	-16 000	-1.605	-1.821	-1.582
Scrub	-20 000	-2.054	-2.166	-1.940
Horticulture	24 000	2.546	2.909	2.766
Exotic Forest	137 000	12.683	15.431	13.177
Ind. Forest	-2000	-0.265	-0.280	-0.248
Total	-3000	-2.770	-1.220	-1.656

The spatial model is giving more weight to the sparse sites, so the larger total C in pasture shown in the spatial model shows that sparse pasture sites tend to have more C than clustered sites. The effect is even stronger in exotic forests, increasing from 92.6 t C ha⁻¹ to 100.1 t C ha⁻¹. A land use where sites are sparse will give about the same estimates for total C as the standard model, since the spatial model will regard them as independent. Estimates of total C in scrub rises from 102.2 t C ha⁻¹ in the spatial model to 104.1 t C ha⁻¹ in the standard model, the small difference suggesting that scrub was sparsely sampled.

The model using only points within 40 km of an exotic forest checks whether the decision to plant a forest could be consistently influenced by a factor related to soil carbon but unrelated to the overall *SoilClim* variable. This factor could be completely independent of soil or climate, or it could be a part of a *SoilClim* by LandUse interaction. Pasture total C is 109 t C ha⁻¹, 2 t C ha⁻¹ lower than in the full model, which explains part the 5.5 t C ha⁻¹ difference from the full model LUE. The remaining 3.6 t C ha⁻¹ must relate to a different mix of the *SoilClim* categories.

2.5.4 Recommendations

The analyses quoted above are indicative only, but they do suggest that the soil carbon loss when exotic forest replaces pasture is overstated by the original linear model. This is potentially a major finding which, if verified, could have very substantial cost implications for New Zealand in terms of liabilities during the first commitment period of the Kyoto Protocol (i.e. liabilities could potentially be reduced by over \$100M during CP1, depending on carbon prices). To advance this work, the following work must be given priority:

- The fit of the exponential correlation model needs to be checked and alternative models tried.
- The differences described here need to be compared with their standard errors to assess the strength of evidence for them.
- Since the estimation of the original model more data has been collected. In particular, data from a recent random survey of scrub and indigenous forest would provide a standard against which the spatial correlation in current data could be assessed.
- It has been previously suggested that the interaction between land use and *SoilClim* should be re-examined (Baisden et al 2006). The analysis of sites close to exotic forests performed as part of this study confirms this suggestion.
- It has also been suggested that a multiplicative model might fit the data better than an additive model, and be more physically realistic (Baisden et al 2006). An informal examination of residuals to the models discussed here reinforces this suggestion.
- A reassessment of the model would provide an opportunity to incorporate all new data.

Overall, this exercise emphasises the difficulty of forming unambiguous estimates from historic data. A properly designed random survey would provide unambiguous, unbiased estimates. Although expensive, the cost of such a survey needs to be set against what increasingly appears to be an excessive liability likely to arise from over-estimates of the loss of soil carbon with afforestation/reforestation made by relying, to date, on estimates made using existing data that have not specifically been collected for the purpose of estimating stock–change factors.

2.6 Mitigation Opportunities for Soil Carbon

2.6.1 Implications of afforestation/reforestation for erosion reduction on marginal lands

The principle mitigation opportunity provided by afforestation/reforestation in relation to soil carbon is the control of erosion through conversion of marginal pasture lands to permanent forests. Many studies have shown that erosion and sediment production from steep pasture-lands prone to soil slip erosion are greatly reduced by the presence of a mature, closed-canopy forest cover (exotic or indigenous). These reductions are attributable to the soil-strengthening ability of root systems, and to the influence of trees on the hydrology of forested slopes (e.g., Pearce et al. 1986; Dymond & Betts, submitted). Mature forests can reduce soil slip erosion rates by 90% compared with equivalent pasture areas (Hicks &

Harmsworth 1989; Marden et al. 1995; Marden & Rowan 1993). During severe rainstorms, such as experienced during Cyclone Bola, up to 30% of land areas may be affected by soil slip erosion, equivalent to soil carbon losses of about 20 t C ha⁻¹. At least as concerning is that it takes many decades, at the very least, for soil carbon stocks to recover and restore land productivity. There is also the fact that the risk of high intensity rainstorms is increasing with climate change.

Forestry thus offers both a more sustainable land use than pastoral farming for steep hill-country, in terms of reducing soil carbon losses, preserving the soil resource, and reducing downstream flood risk and impacts on fisheries. There is some risk of enhanced erosion and sediment generation compared with a pasture cover at the time of forest harvest, every 27–30 years, but soil losses are expected to be substantially less over a typical forest rotation cycle than if the land remained in pasture (Fahey & Coker 2002).

At the same time, while it can easily be shown that erosion removes large amounts of soil carbon from affected sites, the overall net balance of carbon flows to the atmosphere is more complicated than would be assumed by considering only erosion-affected sites. Depending on the fate of carbon after it has been eroded and the rate of recovery of erosion scars, erosion may not necessarily cause a net loss of carbon to the atmosphere. These issues are explored at greater depth in Section 7.

2.6.2 Improving stock-change factors to account for the effect of erosion

To better account for the effects of erosion on the national soil carbon inventory, including accounting for possible gains in soil carbon under forests planted on marginal pasture lands, it is necessary to develop two new stock-change factors that quantify:

- the reduction in soil carbon stocks between undisturbed and erosion-prone pastoral hill country (possibly differentiated by slope class and lithology)
- the difference in soil carbon stocks between forested and pastoral erosion-prone land.

Developing these stock-change factors would also remove what is a weakness in the present Soil CMS model: the current Soil CMS predictive equation inherently assumes the change in soil carbon stocks with forestation is independent of the initial carbon stock under pasture. This seems biophysically unlikely, and a preferred approach would be to have separate stock-change factors for areas likely and unlikely to be affected by soil slip erosion.

It is a matter of some urgency that these new stock-change factors be developed. It is recommended that an intensive grid-based sampling programme be undertaken to determine average soil carbon stocks under forest and grassland in hill country. On areas that otherwise have the same soil type, slope and climate, it may require as many as 500 point samples be taken to at least 30 cm depth for each land cover. This is largely a paired site exercise, but extended to cover a substantially larger area than would normally be sampled, and needs to be done without any restrictions on the soil profiles sampled.

2.7 Current Best Estimates of Forest Soil Carbon Stocks and Change

2.7.1 Carbon stocks and change

Stocks and change under current land use/management (1990–2020)

Overall, the foregoing sections show that definitive data on the magnitude and rate of change in soil carbon stocks with afforestation/reforestation of pastures have yet to be obtained. The following points summarising the available national and international evidence:

- (i) With afforestation/reforestation of pasture lands, there are usually changes in soil carbon stocks. They are small compared to changes in biomass carbon stocks but nonetheless significant. Measured changes under second rotation forests appear to show that further change are negligible, but some modelling suggests smaller losses than under first rotation conditions may continue.
- (ii) For *P. radiata* forests, analysis of national soils datasets using the Soil CMS approach suggests these losses amount to an average 18 t C ha^{-1} , whereas paired site studies suggest an average loss of 8.4 t C ha^{-1} , although given uncertainties associated with these values they are not statistically different.
- (iii) Re-analysis of the Soils CMS dataset to take account of possible spatial autocorrelation in the data suggests that the value of 18 t C ha^{-1} is likely to reduce to about 8 or 13 t C ha^{-1} , depending on the statistical model employed. This further reconciles the different estimates from the CMS and paired sites. This is a very important, but still preliminary result, and was not able to be confirmed using independent data during this study.
- (iv) A meta-analysis of existing studies, both national and international, suggests soil carbon losses with afforestation/reforestation increase with increasing rainfall. At rainfalls typical of those under which New Zealand *P. radiata* forests are planted, mean losses of about 10 t C ha^{-1} appear likely—a value not dissimilar to the mean value for the paired site studies nor to the preliminary result from re-analysis of the Soil CMS dataset.
- (v) Modelling shows that reductions in soil carbon following forestation of grasslands should be expected, and that the magnitude of the loss can be broadly explained by a increase in nitrogen storage in woody litter and biomass with only slight changes in the soil C:N ratio, together with increased leaching at higher rainfall. Losses of order 10 t C ha^{-1} seem likely for the conditions under which many *P. radiata* forests are grown in New Zealand.
- (vi) It cannot yet be determined whether the change in soil carbon stocks follows a quasi-linear, sigmoidal or exponential time course: different modelling studies presently indicate that various options are consistent with the available data. More complex time sequential changes may also occur due to time-varying interactions between carbon and nitrogen stocks, rainfall, stand productivity, and litter decomposability.

Overall, it is concluded that:

- the overall weight of evidence from various studies suggests a value for the afforestation/reforestation soil carbon stock-change factor of about 10 t C ha^{-1} ,

occurring over a period of c. 20 years. A loss of this magnitude is similar to the gain in carbon stocks in forest floor fine litter;

- it remains unclear whether soil-carbon losses continue at a lesser rate under second rotation forests, or whether stocks remain static;
- there currently is no compelling evidence to reject the IPCC default of a constant decay rate over a 20-year period as the best estimate of the average behaviour;
- that after deforestation, it is likely that soil carbon stocks recover to pre-forestation values given that deforestation in New Zealand almost always involves a land-use change to managed pastoral farming.

Under the Kyoto Protocol, all carbon pools that show net losses during a commitment period must be accounted. For national accounting, changes in the soil carbon pool of forests planted after 31 December 1989 will therefore have to be estimated. However, it should be noted that changes in the soil carbon pool with planting of *P. radiata* are likely to be very similar to gains in forest floor fine litter stocks from needles and small twig/branch mortality—of about 12 t C ha⁻¹ (Scott et al. 2006b; Kirschbaum et al. 2008b). A similar balance between soil carbon losses and gains in fine litter appears likely for shrubland (Tate et al. 2003; Trotter et al. 2005). Domestic accounting schemes, such as the Emissions Trading Scheme (ETS), or the Permanent Forest Sinks Initiative (PFSI), could for simplicity neglect accounting of both the soil carbon and fine litter pools without any significant overall loss in environmental integrity (as judged by total net emissions to the atmosphere). This would be approximately true during both the afforestation/reforestation, and deforestation phases.

Stocks and change for post-2012 mitigation options

The evidence is very clear that establishing a forest cover on erodible pastoral hill country results in a major reduction (c. > 90%) in shallow soil slip erosion caused by high intensity rain storms. The resultant gains in soil carbon—which may be of order 20–30 t C ha⁻¹ in the medium to long term (50–100 years; K. Tate pers. comm.)—could be accounted under net-net or full-carbon-accounting approaches (including under land management options available under Article 3.4 of the Kyoto Protocol).

However, adoption of either of those accounting options would greatly reduce forest-sink biomass offsets that could be accounted by New Zealand. Possible gains from inclusion of carbon gains on erodible hill country would be much more modest in comparison.

2.7.2 Discussion

Implications of forecasts and scenarios

It can be expected that under future afforestation/reforestation programmes such as the ETS or PFSI aimed at expanding total forest area, there will be small but significant changes in mineral soil carbon. On a “weight of evidence” basis, losses of about 10 t C ha⁻¹ seem likely for many areas where *P. radiata* is planted. These losses are expected to be largely offset by accumulation of fine litter on the forest floor, which might occur over similar timeframes. Deforestation, with conversion of forest to managed grasslands, is likely to reverse both the soil carbon losses and litter carbon gains.

Effects of information gaps/uncertainties on forecast/scenario reliability

There is still no well-justified carbon stock-change factor for afforestation/reforestation in New Zealand, notwithstanding the “weight of evidence” argument advanced above. However, the stock change factor is unlikely to change substantially from the currently estimated value of 10 t C ha^{-1} , provided the results of the re-analysis of the Soil CMS calibration dataset are considered to be valid upon further testing against independent data. There is also a need to develop stock-change factors for erodible hill-country lands, in order to make overall estimates of soil carbon stocks and stock-change more accurate, defensible and transparent.

Without better-developed stock-change factors, the impact of afforestation/reforestation on soil carbon will continue to remain uncertain, although any changes in soil carbon stocks will still be small in comparison with forest biomass carbon stocks. Once new soil carbon stock-change factors have been developed for erodible hill-country, it might become possible to include likely soil carbon gains with afforestation/reforestation on such lands.

2.8 Conclusions and Recommendations

2.8.1 Present status of studies, datasets, analyses and forecasts

New Zealand’s Carbon Monitoring System had developed a best estimate of change in soil carbon stocks with afforestation/reforestation, supported by peer-reviewed scientific publications, as a loss of 18 t C ha^{-1} . Given there are about 650 000 ha of Kyoto exotic forests, at $\$25 \text{ t CO}_2$ this equates to a liability of about $\$270\text{M}$ over CP1.

A summary of paired-site studies concluded that afforestation/reforestation would result in a loss of only 8.5 t C ha^{-1} , which would reduce New Zealand’s liability from soil C losses to $\$125\text{M}$ (using the same assumptions of carbon costs, etc., as above).

Above, we have presented a re-analysis of the original data that takes the spatial auto-correlation between different measurements into account. Two different approaches have been used to deal with the effect of auto-correlation in the data. They result in revised CMS estimates of carbon losses of 8 or 13 t C ha^{-1} , which would place the liability from soil-carbon losses at $\$120$ or $\$190\text{M}$.

All these estimates have considerable statistical uncertainty bounds so that even the most divergent estimates do not differ statistically significantly from each other. So, while the most recent available evidence suggests that the most likely carbon loss is about 10 C ha^{-1} , the variability in the data means that losses substantially higher or lower than 10 C ha^{-1} cannot yet be excluded.

Clearly, insufficient work has been done to date to quantify this figure better and reduce the attendant uncertainty. The effort devoted to this task is thus totally disproportionate to the magnitude of the potential cost imposed by soil carbon losses with afforestation/reforestation. Given the weight of evidence, it seems very desirable for a substantial effort to be made between now and the end of CP1 to better refine the likely C loss. It is likely that this would yield an excellent return on that investment: if it is able to be transparently and defensibly justified that a figure of even 11, rather than 18 t C ha^{-1} is the appropriate stock-change figure, this would represent a $\$100\text{M}$ saving.

Further substantial gains would be possible if the time-dependence of the carbon stock change could be proved to have a sigmoidal or exponential, concave-downwards shape, with fewer losses in later rather than earlier years after the land-use change due to the exotic forest planting boom in the early 1990s. However, there is less confidence about whether a favourable outcome is likely for studies to better define the time dependence of the stock-change factor.

Overall, there is considerable urgency to conduct a much more comprehensive programme of work to quantify changes in soil carbon stocks with afforestation/reforestation, including determining/validating stock-change factors for erosion-prone lands.

2.8.2 Key uncertainties, information gaps and research priorities

Key uncertainties have been largely dealt with in the last section above, but in summary are:

- The carbon stock-change factor for afforestation/reforestation in New Zealand remains inadequately defined.
- It is not known if the factor is appropriate for erodible hill-country lands.
- The time-dependence of carbon stock change with afforestation/reforestation remains poorly defined.
- Present conclusions on the magnitude of stock-change factors rely on space-for-time substituted experiments, due to an absence of long-term studies.
- There are very few data with which to determine the time dependence of carbon-stock changes, and as such, uncertainty about the trajectory of soil carbon change is high.

Considered overall, these uncertainties are not expected to have a large effect on medium- to long-term forecasts of the magnitude of net emissions or removals, because it is expected that the removals and emissions components will tend to cancel each other out. However, the certainty and defensibility with which this conclusion can be formally stated, and its transparency under international review, presently remain limited. Neither is it possible to determine at present whether New Zealand's liabilities from the change in forest soil carbon stocks over CPI might be able to be minimised because the time dependence of change is a downwards concave exponential.

There is thus a pressing need for research in the following areas:

- Confirm whether analysis of the Soil CMS dataset supports the initial conclusion that taking account of spatial autocorrelation in the data is likely to result in a substantially smaller stock-change factor for afforestation/reforestation than presently used. This conclusion could not be confirmed as part of the present study because the Ministry for the Environment was not prepared to release national inventory data.
- Assemble a definitive, representative dataset of average litter stocks in Kyoto exotic forests to confirm whether carbon stocks in forest floor litter are approximately equal to (revised) figures for soil carbon losses with afforestation/reforestation. If that can be

done, it would greatly simplify the accounting in domestic initiatives such as the PFSI and ETS.

- Conduct paired-grid studies (i.e. much more intensive, grid-based sampling than in paired-site studies) to determine reliable stock-change factors for erosion-prone hill country for pasture and forested areas. It is expected these factors will decrease New Zealand's Kyoto liabilities and may advance the use of forests in erosion-prone hill country as a carbon loss mitigation strategy that would be accountable under the Kyoto Protocol.
- Conduct modelling studies to further clarify the likely time dependence of soil carbon change with afforestation/reforestation using the best New Zealand data, and based on a fully coupled C-N dynamics model (the use of CenW is strongly recommended).
- Establish long-term monitored sites on erosion-prone and non-erosion-prone sites (for several key soil types) to generate definitive datasets of the magnitude and time dependence of soil carbon change with afforestation/reforestation.

2.8.3 Implications of accounting and mitigation options for New Zealand's post-2012 net position

Regardless of the particular post-2012 accounting regime that is finally endorsed internationally, changes in carbon stocks with afforestation/reforestation will have to be quantified and accounted. Under gross-net accounting approaches—that New Zealand will argue strongly for—there is a definite advantage in minimising accountable soil carbon losses.

If net-net accounting schemes of whatever kind (all lands, or involving some subset of managed lands) were adopted, it would actually be advantageous if the existing large stock-change factor of 18 t C ha^{-1} was to remain, as this would increase total 1990 baseline emissions, whereas future soil-C losses would trend towards zero as more and more forest reach an age beyond the assumed 20 years for adjustment in soil C stocks.

However, net-net accounting in any form would carry so many other major disadvantages for New Zealand that the issue of soil carbon change would be of relatively minor importance. For this reason, it presently seems very unlikely that New Zealand would support initiatives for the adoption of net-net accounting, although the implications for New Zealand of Canada's forward-looking (net-net) baseline proposal have yet to be fully determined.

Assuming that international negotiations finally settle on a LULUCF accounting framework that is not dissimilar to that in CP1, perhaps with mandatory but electively-capped Article 3.4 activities, it will continue to be important to minimise liabilities from change in soil carbon stocks with afforestation/reforestation. As the government is introducing a range of new incentives for establishment of at least 20 000 ha of new forests annually to contribute to post-2012 emissions offsets, accounting of soil carbon losses with afforestation/reforestation is likely to continue to be required.

3. Biochar Amendment

Troy Baisden (GNS)

3.1 Introduction

Considerable literature has emerged in the last 10 years on the importance of black carbon (BC) in the global carbon cycle (Schmidt et al. 2000; Lehman 2007). This includes recognition of BC derived from fossil sources as well as the charring of plant material, which may create potential for a C sink (Lehman 2007). Here, biochar is considered synonymous with BC because charring of biological material is the main source of BC in most soils, and because fossil and other combustion sources of BC are not expected to be significant in a New Zealand context.

3.1.1 BC issues relevant to BC accounting, monitoring and verification

Two primary difficulties govern the development of accounting, monitoring and verification methodologies for BC. First, the nature of BC is paradoxical: the main chemical structures of BC are highly resistant to decomposition in soil, yet a significant proportion of BC added to soils appears to undergo dynamic changes in chemistry, including oxidation (Czimzik & Masiello 2007). Approximately 20% of char-derived BC stocks have been observed to disappear over 100 years (Hammes et al. 2008). The paradoxical nature of BC as both dynamic and inert is considered further in the development of mass balance equations for BC accounting below. Second, in contrast to total soil organic carbon (TSOC), there is not yet a single standard method for the quantitative analysis of BC (Hammes et al. 2007). Methodologies for measuring BC are not considered further here, but present a challenge for monitoring and verification due, in part, to the likelihood that considerable BC already exists in many soils.

In this report, primary focus is placed on the development of mass balance equations to provide a robust basis for C accounting relevant to New Zealand. The approach taken captures the current state of knowledge describing both dynamic and highly resistant fractions of biochar. The chemical properties of these fractions are discussed further, but it is important to note that beneficial soil properties associated with biochar may largely be associated with the dynamic fraction (Lehman 2007).

3.1.2 Background BC concentration required for BC monitoring and verification

To our knowledge, few, if any, measurements of black carbon in New Zealand soils have been published using any of the suite of methods recently recognised as appropriate for BC quantification (Hammes et al. 2007). It would therefore be inappropriate to assume that New Zealand soils lack BC so that any additions of BC could be monitored and verified against a negligible background concentration. Indeed, it is reasonable to expect that Polynesian burning as well as the fires associated with European land clearing produced significant quantities of BC.

FRST-funded collaboration between SLURI, GNS and CSIRO has recently completed MIR analysis of a suite of 519 New Zealand soil horizons. This allows the PLS predictions of BC developed for Australian soils to be applied to New Zealand soils. The results should be

considered preliminary because Australian PLS calibrations are used without any New Zealand calibration datasets, for which NMR measurement are about to commence. Due to the preliminary nature of the data from MIR, Table 1 presents the median and upper quartile of the results in order to conservatively estimate the degree to which biochar additions are likely to be detectable above background concentrations of BC. The results suggest New Zealand

soils may commonly have BC concentrations of 0.4% or more, with the BC pool representing at least 14% of total soil organic carbon (SOC) in many soils. The measurements support the suggestion that there is a significant and measurable background level of BC in New Zealand soils that needs to be understood before efforts to monitor and verify the fate of BC additions can be undertaken with confidence.

Table 1 Preliminary data from CSIRO MIR analysis of 519 New Zealand soil horizons

	All 519 New Zealand soil horizons	235 NZ soils horizons best matched to Australian calibration data
Char – Median (g/100g)	0.23 g/100 g	0.22 g/100 g
Char – Upper Quartile	0.41 g/100 g	0.40 g/100 g
Char as % of Total Organic Carbon – Median	10.5%	8.3%
Char as % of Total Organic Carbon – Upper Quartile	23.2%	13.7%

3.2 Deriving a system of equations for net C implications of biochar

With the goal of developing a formula for the net C implications of biochar production and application to soil, it is necessary to develop a system of equations that describe all relevant processes affecting the mass balance of C. The equations derived here take full account of the available scientific literature on the fate of biochar in soil, as well as the fact that biochar production will divert organic residues which otherwise would be applied to, or incorporated into, the soil. By deriving a system of differential equations, the mass balance of C is fully described. In all cases presented here, analytical solutions to the equations have been obtained as a function of time (using the Matlab symbolic toolbox). The resulting effect of biochar production on atmospheric CO₂ can then be summarised as a function of time for relevant scenarios.

3.2.1 Equations for continuous biochar inputs (cropping)

The continuous equations below are most appropriate for a situation where biochar inputs occur every year for a long period. A one-off input will be examined later as a separate case to describe a typical application of biochar to pastures.

$$\frac{dR}{dt} = (1 - f_{BR} - f_{BS})R_{in} - (k_{RS} + k_{Rox})R \quad 1a$$

$$\frac{dS}{dt} = k_{RS}R - k_{Sox}S \quad 1b$$

$$\frac{dI}{dt} = 0 \quad 1c$$

$$\frac{dB_R}{dt} = \varepsilon(1 - f_I)f_{BR}R_{in} - c_B k_{Rox}B_R \quad 1d$$

$$\frac{dB_S}{dt} = \varepsilon(1 - f_I)f_{BS}R_{in} - c_B k_{Sox}B_S \quad 1e$$

$$\frac{dB_I}{dt} = \varepsilon f_I (f_{BS} + f_{BR})R_{in} \quad 1f$$

The purpose of these equations is to describe the input and outputs affecting each relevant pool of soil C. Six pools are described, including three pools for general soil C and three pools explicitly created to represent biochar additions. The three pools for general soil C are plant residues, R , stabilized soil C, S , and inert (or passive) soil C, I . These pools have sub-annual, decadal, and millennial residence times respectively, and correspond to the structure of most process-based soil C models such as RothC and CENTURY (Baisden & Amundson, 2003). This approach allows parameters describing R , R , and I to be derived from New Zealand measurements rather than process models developed overseas. Biochar additions are accounted for by designating three pools with essentially identical function as the three main soil pools for residue, stabilized and inert soil C, termed B_R , B_S , and B_I .

In equation 1a, the input of C to soil in these equations occurs as inputs to residue, R_{in} . The diversion to biochar of a fraction of residue that would have been returned to soil is represented by the terms f_{BR} and f_{BS} , which are assumed to be added as surface residue or incorporated into stabilised soil aggregates, respectively. These two fractions allow consideration of surface applied biochar versus biochar incorporated into the soil. Surface biochar may be preferentially exposed to additional loss processes including surface erosion (Rumpel et al. 2006), oxidation by sunlight or atmospheric ozone (Czimczik & Masiello 2007). In contrast, biochar incorporated into the soil will not be exposed to these processes and will receive additional protection from decomposition afforded by soil mineral surfaces and aggregation (Czimczik & Masiello 2007). The fractions of inputs described by f_{BR} and f_{BS} are routed at inputs to the biochar residue and stabilised pools, B_R and B_S , respectively. These inputs are therefore subtracted from residue inputs (R_{in}) that would have been added to soil C. They are instead added to the biochar pools, but only after taking account of the fact that only approximately 65% of the original residue C is recovered from the pyrolysis process (Lehman 2007), using the efficiency term ε (eqs 1d–f).

Plant residues not converted to biochar enter the residue pool, R , and are either oxidised at the rate k_{Rox} , or stabilised in soil at that rate k_{RS} . When they have entered the stabilised soil pool, S , they are assumed to decompose via oxidation to CO_2 at the rate k_{Sox} (eq. 1b). Due to its millennial residence times, the inert pool, I , is assumed not to change for the purpose of this work, and therefore has neither inputs nor outputs (eq. 1c).

The biochar pools, B_R and B_S , have identical decomposition dynamics to the main soil C pools, R and S , except that decomposition of biochar is assumed to be retarded by a factor, c_B

relative to general soil C decomposition (eqs 1d, 1e). Based on observations made on historic BC sources and in the laboratory, a retardation factor of 6 (range 4–9) appears to be supported (Cheng et al. 2008a, b). This retardation factor is probably most appropriately applied to BC in the stabilised soil pool. However, it has been assumed that BC enters the residue pool as a most pessimistic scenario. The main effect of such a scenario is to limit the C sequestration benefits of BC addition largely to the pool of inert biochar, B_I . Inert biochar is formed and assumed not to decompose (equation 1f; Lehman 2007; Czimczik & Massiello 2007). Based mainly on observations obtained on soils sampled over 100 years after prairie burning ceased (Hammes et al. 2008), inert biochar is assumed to comprise 70% (range 60–90%) of BC as defined by f_I , with the remainder having the potential to decompose.

For modelling purposes, the following initial conditions are used. First, $S + I$ is set to the default C level specified in Tate et al. (2005). For cropping soils appropriate to receive continues inputs of charred residues, this level is 97 tC/ha to a depth of 30 cm. Based on several New Zealand studies (Baisden, unpublished; Prior et al. 2007; Baisden & Parfitt 2007), the inert pools has been estimated to be approximately 26–35% of the total soil C pool. A value of 35% has been used for this work. The biochar pools B_R , B_S and B_I are assumed to start at zero, while the residue pool, R , is allowed to achieve a value along with inputs, R_{in} , to establish a steady state system. Because the purpose of these equations is to model the fate of biochar inputs, the detail required to fully model the ephemeral residue pool has not been included in the model presented here. As a result, the C stock in the residue pool is not considered in providing estimates of the net C balance effect of biochar production and addition to soil. This is expected to have a negligible effect on the net C balance under most if not all situations.

Thus, for cropping systems where a proportion of residues are processed to biochar and added to the soil, it is possible to assign parameter values for best estimate, pessimistic and optimistic scenarios as given in Table 2.

Table 2 BC parameters for continue BC conversion from cropping residues. All stocks in tC/ha are to 0.3 m soil depth

Parameter	Best Estimate	Pessimistic	Optimistic	Units
f_{BR}	0	0	0	-
f_{BS}	0.2	0.1	0.4	-
k_{RS}	2	2	2	yr ⁻¹
k_{Rox}	2	2	2	yr ⁻¹
k_{Sox}	0.06	0.02	0.1	yr ⁻¹
f_I	0.7	0.6	0.9	
R_{in}	7.5	2.4	12.0	-
c_B	0.17	0.25	0.11	-
k_{kill}	0	0	0	tC/ha
S_0	59.9	59.9	59.9	tC/ha
R_0	1.9	0.6	3.0	tC/ha
B_{R0}	0			tC/ha
B_{S0}	0			tC/ha
B_{I0}	0			tC/ha
ϵ	0.65	0.65	0.65	-

When the parameters shown in Table 2 are applied to equations 1a–f, the resulting estimate of net C balance following biochar production and addition to soil is shown in Table 3.

Table 3 Net C balance scenarios from continuous biochar production and addition to cropland (tC/ha change relative to initial condition)

Year	0	1	5	10	20	50	100
Best Estimate	0.0	0.4	1.8	4.1	10.4	34.0	74.3
Pessimistic	0.0	0.1	0.2	0.5	1.1	3.6	9.1
Optimistic	0.0	1.4	6.5	16.1	41.0	128.2	275.0

The values in Table 3 suggest a very large range of plausible estimates for potential C sequestration resulting from biochar production and soil incorporation in New Zealand croplands. Perhaps surprisingly, the parameters chosen for biochar production and stabilisation have a relatively small influence on the total range. In contrast, much of the total range relates to the total production of available residue (R_{in}) and the proportion of that residue that is diverted to biochar (f_{BS}). These two factors explain essentially all of the variation shown in best estimate, pessimistic and optimistic net sequestration rates for year 1. In subsequent years, however, variables related to soil C dynamics become increasingly influential reflecting both the fate of the biochar, and the fate of the residue that is diverted to biochar rather than incorporated into the soil. It is important to note that, due to the structure of the model, some of this uncertainty in residue dynamics is related to the amount of residue that contributes to the stabilisation of soil C, rather than total residue.

3.2.2 One-off inputs of biochar to pastures

Given the large land areas of pastoral land in New Zealand compared with the availability of residues for biochar production, for pastures, a more realistic scenario may be one-off applications of biochar. This might occur, for instance, in combination with pasture renewal. In such a case, we would have a one-time routing of biochar into the soil pools above. This can be calculated using the input terms only, and neglecting decomposition terms. However, tillage may be required for biochar to be incorporated into the soil and that will cause some soil C loss, which we can represent by the parameter k_{till} and assign a value of 6% (range 3–11%) of the soil C stock (Conant et al. 2007). It must be noted, however, that in some circumstances, biochar could be incorporated through tillage that would have occurred anyway—such as commonly occurs during pasture renewal. In such a case, the tillage loss (k_{till}) can be set to zero, but this approach has not been favoured here, even as an optimistic case, because no information of tillage practices is available. Results in the absence of tillage can be provided, however, to be used when it is justifiable to assume tillage would have occurred in the absence of biochar incorporation.

Since biochar inputs and tillage losses can be assumed to occur instantaneously, we can remove the time factor from the equations above, and write the system of equations as a difference relative to the prior state of the system.

$$R_1 = R_0 \quad 2a$$

$$S_1 = -k_{ill}S_0 + S_0 \quad 2b$$

$$I_1 = I_0 \quad 2c$$

$$B_{R_1} = (1 - f_I)f_{BR}R_{Bin} + B_{R_0} \quad 2d$$

$$B_{S_1} = (1 - f_I)f_{BS}R_{Bin} + B_{S_0} \quad 2e$$

$$B_{I_1} = f_I(f_{BS} + f_{BR})R_{Bin} + B_{I_0} \quad 2f$$

In these equations, terms subscripted zero refer to the prior state and terms subscripted I refer to the new condition immediately after biochar addition and incorporation. The parameter R_{Bin} refers to the quantity of biochar added. In this case, it is assumed that R_{Bin} is the actual C content of the biochar, which has already been converted from biomass with an efficiency ϵ , as defined in equation 1a–f.

The new conditions defined by eqs 2 can then be used as the initial condition to determine net C balance over a period of years. The resulting system of equations (eqs 3), starting with initial conditions of R_I , S_I , I_I , B_{R_I} , B_{S_I} , B_{I_I} , becomes a simpler version of equations 1a–f:

$$\frac{dR}{dt} = R_{in} - (k_{RS} + k_{Rox})R \quad 3a$$

$$\frac{dS}{dt} = k_{RS}R - k_{Sox}S \quad 3b$$

$$\frac{dI}{dt} = 0 \quad 3c$$

$$\frac{dB_R}{dt} = -c_B k_{Rox} B_R \quad 3d$$

$$\frac{dB_S}{dt} = -c_B k_{Sox} B_S \quad 3e$$

$$\frac{dB_I}{dt} = 0 \quad 3f$$

As above, $S_0 + I_0$ is set to the default C level specified in Tate et al. (2005), which is 109 tC/ha for pastures. Similarly, New Zealand studies (Baisden unpublished data; Prior et al. 2007; Baisden & Parfitt 2007) allow the inert pool to be estimated as approximately 26–35% of the total soil C pool. A value of 35% has been used for this work. The biochar pools B_R , B_S and B_I are assumed to start at zero, while the residue pool, R , is allowed to achieve a value along with inputs, R_{in} , to establish a steady state system. Because the highly ephemeral residue pool is not represented in detail, the C stock in the residue pool is not considered in providing estimates of the net C balance effect of biochar production and addition to soil.

Thus, for pasture soils where biochar produced off-site may be added to soil, it is possible to assign parameter values for best estimate, pessimistic and optimistic scenarios as indicated in Table 4.

Table 4 BC parameters for one-off addition of BC to pasture soils. All stocks in tC/ha are to 0.3 m soil depth

Parameter	Best Estimate	Pessimistic	Optimistic
f_{BR}	0	1	0
f_{BS}	1	0	1.0
k_{RS}	2	2	2.0
k_{Rox}	2	2	2.0
k_{Sox}	0.0625	0.1	0.02
f_I	0.7	0.6	0.9
R_{in}	8.9	14.2	2.8
R_{Bin}	3.2	1	1.0
c_B	0.17	0.25	0.11
k_{ill}	0.06	0.11	0.03
S_0	70.9	70.9	70.9
R_0	2.2	3.5	0.7
B_{RO}	0	0	0
B_{SO}	0	0	0
B_{IO}	0	0	0

When the parameters shown in Table 4 are applied to eqs 3a–f, the resulting estimates of net C balance following biochar production and addition to soil are shown in Table 5.

Table 5 Net C balance scenarios from the one-off production and addition of 1 and 10 tC/ha to pasture. Values reported are in tC/ha change relative to prior state. Estimates preceded by NT assume tillage would have occurred anyway and therefore ignore tillage-induced C loss by setting $k_{ill}=0$

Year	0	1	5	10	20	50	100
Best Estimate							
1 t addition	-3.3	-3.0	-2.1	-1.3	-0.3	0.7	0.8
10 t addition	5.7	6.0	6.7	7.4	8.2	8.6	8.1
NT 1 t addition	1.0	1.0	1.0	1.0	0.9	0.9	0.8
NT 10 t addition	10.0	10.0	9.8	9.7	9.4	8.8	8.1
Pessimistic							
1 t addition	-6.8	-6.2	-4.1	-2.2	-0.4	0.6	0.6
10 t addition	-0.5	2.7	2.9	4.5	6.3	7.3	7.4
NT 1 t addition	1.0	0.8	0.6	0.6	0.6	0.6	0.6
NT 10 t addition	10.0	8.4	6.3	6.1	6.0	6.0	6.0
Optimistic							
1 t addition	-1.1	-1.1	-0.9	-0.7	-0.4	0.2	0.7
10 t addition	7.9	7.9	8.1	8.2	8.5	9.1	9.5
NT 1 t addition	1.0	1.0	1.0	1.0	1.0	1.0	1.0
NT 10 t addition	10.0	10.0	10.0	10.0	10.0	9.9	9.8

Main features of the results presented in Table 5 include the following points:

- Tillage has an important effect in all scenarios, but the effect of tillage diminishes over time assuming further tillage does not occur.
- Small (1 tC/ha) additions of biochar lead to significant C emissions to the atmosphere after tillage is accounted for.
- The oxidation rate of soil C, k_{Sox} , has an important effect on the longevity of the loss due to tillage, and therefore has potential implications for identifying an interval between biochar addition events that avoids soil C losses due to tillage incorporation.
- The more rapid oxidation of dynamic fraction of biochar under the pessimistic scenario has a pronounced effect within 5 years.
- The proportion of biochar assumed to be inert (f_I) can have a pronounced effect in the long term, and possibly in the short term, depending on the residence time assumed for the dynamic fraction of biochar (pessimistic scenario).
- Overall, the results show net sequestration over long periods (e.g., 100 years) but a much broader range of possibilities—including net emissions to the atmosphere—over likely accounting periods (e.g., 5 years).

3.2.3 Accounting for the fate of diverted biomass residue

The estimates in Table 5 do not take account of the fate of the biomass residues that would have been used to produce biochar. Accounting for these residues is important if a significant proportion of this residue would not have been oxidised to CO₂ during the accounting period (e.g., 5 years). A range of approaches could be specified where data are available. A simple but realistic approach is provided here. Assuming biochar was produced from forestry waste that otherwise would have lain on the forest floor as coarse woody debris (CWD), the fate of this diverted C should also be accounted for. This is accomplished by integrating the simple equation,

$$\frac{dR}{dt} = -k_{CWD}R; R_0 = \frac{R_{Bin}}{\epsilon} \quad 4$$

With the assumption of a simple first order decomposition with a mean residence time ($1/k_{CWD}$) of 3 years (range 2–5 years), and an initial quantity of residue that takes account of the biochar conversion efficiency, ϵ , then the quantity of C that would have remained as CWD can be added to the net C balances in Table 6.

Table 6 Net C balance scenarios from the one-off production and addition of 1 and 10 tC/ha to pasture, after including accounting for CWD as specified in eq. 4. Values reported are in tC/ha change relative to prior state. Estimates preceded by NT assume tillage would have occurred anyway and therefore ignore tillage-induced C loss by setting $k_{till}=0$

Year	0	1	5	10	20	50	100
Best Estimate							
1 t addition	-4.8	-4.1	-2.4	-1.4	-0.3	0.7	0.8
10 t addition	-9.6	-5.1	3.8	6.9	8.2	8.6	8.1
NT 1 t addition	-0.5	-0.1	0.7	0.9	0.9	0.9	0.8
NT 10 t addition	-5.4	-1.1	6.9	9.2	9.4	8.8	8.1
Pessimistic							
1 t addition	-8.3	-7.5	-4.7	-2.4	-0.4	0.6	0.6
10 t addition	-15.9	-9.9	-2.7	2.4	6.0	7.3	7.4
NT 1 t addition	-0.5	-0.4	0.1	0.4	0.6	0.6	0.6
NT 10 t addition	-5.4	-4.2	0.7	4.0	5.8	6.0	6.0
Optimistic							
1 t addition	-2.7	-2.0	-1.1	-0.8	-0.4	0.2	0.7
10 t addition	-7.5	-1.4	6.8	8.1	8.5	9.1	9.5
NT 1 t addition	-0.5	0.1	0.9	1.0	1.0	1.0	1.0
NT 10 t addition	-5.4	0.7	8.7	9.9	10.0	9.9	9.8

The results presented in Table 6 suggest additional losses of C to the atmosphere over short periods, when compared with those in Table 5. These losses become insignificant after 10–20 years.

3.2.4 A simple formula to approximate net C balance of biochar application to pasture

The values in Table 6 can be reproduced for any time period after BC addition using a simple formula. Assuming a time interval of 5 years is the main period of interest, a formula for the net C balance (NCB_5) after 5 years is as follows.

$$NCB_5 = R_{Bin} \left(BCloss_5 - \frac{CWDfate_5}{\epsilon} \right) - TillLoss_5 \quad 5$$

In this formula, R_{Bin} is the quantity of C in biochar applied to soil, as defined above. $BCloss_5$ is the proportion of BC oxidised during the 5 years since incorporation. $CWDfate_5$ is the proportion of diverted CWD that would remain after 5 years, and the production of biochar is accounted for using the conversion efficiency, ϵ , as defined previously. Finally, $TillLoss_5$ is the depression in soil C resulting from tillage that is still present after 5 years, noting that there will have been some recovery. A similar expression could be derived for any period (e.g., 10 years, etc.). Values for these parameters are given in Table 7.

Table 7 Parameters for use approximating the 5 year net carbon balance resulting from biochar addition described in eq. 5. The NCB_5 values given assume $R_{Bin} = 10$ tC/ha

	$BCloss_5$	$TillLoss_5$	$CWDfate \epsilon$	NCB_5
Best estimate	0.98	3.1	0.29	3.8
Pessimistic	0.63	3.3	0.57	-2.7
Optimistic	1	1.9	0.13	6.8

3.3 Conclusions

These calculations broadly confirm that there is a substantial potential for C sequestration benefits from biochar incorporation in cropland and pasture soil. Nevertheless, the calculations also suggest considerable uncertainty, with pessimistic calculations giving results over 5 years that would appear to be uneconomic in croplands and a net source of C to the atmosphere for pastures. There is a wide range of values between estimates based on pessimistic and optimistic parameter settings, but in all cases it is clear that over reasonable accounting periods (e.g., 5 years), the net carbon sequestration resulting from biochar addition to soils will be considerably less than the quantity of biochar added.

Therefore, the main value of these calculations is likely to be the introduction of robust equations for accounting for New Zealand conditions that allow areas of major uncertainty to be identified and targeted for future research. For continuous biochar production under cropping, the main identified uncertainties relate to the total quantity of residue available for biochar production. For one-off additions to pastures, main sources of uncertainty relate to accounting for the diverted biomass used to produce biochar, the residence times of soil C and the dynamic fraction of biochar, the proportion of biochar that is effectively resistant to decomposition, and the loss of soil C occurring due to tillage and incorporation.

4. Effects of Forest Management Practices on Soil Carbon

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4.1 Introduction

A range of forest management practices have the potential to influence soil carbon stocks. In this section, the New Zealand studies and datasets relating to these practices are reviewed under the topics of silvicultural practices, forest harvesting, residue management, mechanical site preparation, and site improvement. A brief overview of key findings in the international literature is also given. The review is confined to plantation forests: data on New Zealand's indigenous forest soils are not yet available.

This section begins with a review of New Zealand studies that have mainly been presented in published papers, although a review of some unpublished reports is also included. Key datasets containing information on forest management effects on soil carbon in New Zealand are then reviewed, and a summary description of all relevant datasets is appended. Most of these datasets are derived from existing on-going long-term trials, some of which have been established to encompass a range of sites and soils across New Zealand. A number of the published studies on soil carbon stocks reviewed in the first section present results from these ongoing trials.

In addition to the New Zealand studies, results of some key international studies of management impacts on soil carbon stocks are briefly reviewed. This is mainly with a view to determining whether the results of the New Zealand studies are consistent with international studies and whether the findings of the international research can be applied to New Zealand circumstances. Potential opportunities for mitigating soil carbon loss arising from management practices are then discussed. Estimates of the effects of management practices on soil carbon stocks are presented where relevant data were available. As there is no available information yet available on the areas of plantation forest affected by the different management practices in New Zealand, only the magnitude of stock changes that have occurred under specific site and experimental conditions (from various field studies) can be presented.

A range of site and stand management practices are currently employed within New Zealand's exotic plantation forests. The specific set of practices can vary among forests, depending on site and other environmental conditions, and may have changed over time as knowledge and technologies have improved. Many of these practices can potentially affect soil carbon stocks by changing the quantity or quality of organic matter inputs to the soil, by causing physical disturbance of the soil profile, or by modifying soil moisture and nutrient levels which can affect rates of organic matter decomposition.

4.1.1 Definition and overview of management groups

Previous reviews have grouped specific forest management practices in a variety of ways in order to more effectively consider the impacts of these practices on soil carbon stocks (Johnson 1992; Johnson & Curtis 2001; Jandl et al. 2007). For the purposes of this review, five forest management practice groups, encompassing all key forest management practices

in New Zealand, have been identified: (1) silvicultural practices, (2) forest harvesting, (3) residue management, (4) mechanical site preparation, and (5) site improvement. The groups are defined below.

Silvicultural practices

The silvicultural practices group includes species selection, tree stocking rates, thinning and pruning, all of which have the potential to influence the quantity and quality of organic matter inputs to the forest floor and, ultimately, the mineral soil. Soil carbon may also change over time simply with stand development over the course of a rotation. These temporal changes could be assessed by comparing soils under stands of different age classes. Therefore, any studies or datasets relating to the effect of age class are also included in this management group.

Forest harvesting

The forest harvesting group encompasses the different operations that might be used to harvest trees (e.g., hauler-based versus ground-based operations). Different harvesting operations may cause different levels of soil disturbance, compaction, and incorporation of organic residues (e.g., mixing to variable depths).

Residue management

After harvesting, there are several different management practices used for dealing with the harvest residues and forest floor materials, such as:

- retaining residues on-site and allowing them to decompose (either spread across the site or piled into windrows);
- retaining residues on-site and burning them; or
- removing the residues for some other purpose (e.g., fuel for electricity co-generation).

These practices, which have the potential to affect the amount, type, and spatial distribution of organic matter inputs to the mineral soil, are encompassed by the residue management group.

Mechanical site preparation

Mechanical site preparation includes practices that physically prepare the soil and site for replanting by clearing patches of slash (harvest residues) from the ground surface and cultivating the soil, or by using various techniques such as spot-mounding or line-ripping to create conditions conducive to seedling survival and vigorous growth (i.e. a slightly elevated position with free-draining, friable soil). The soil is invariably physically disturbed to some degree during mechanical site preparation which can affect soil carbon stocks.

Site improvement

Other management practices that may either be applied as part of general site preparation or post-planting site maintenance include the application of various fertilisers (most commonly N and P fertilisers), and weed control via the use of herbicides or by other means. Alteration of soil nutrient levels through fertiliser application, particularly in the case of N, could have implications for the levels and dynamics of carbon in the soil whereas the removal of weeds (vegetative ground cover) might influence mineral soil organic matter inputs and moisture regimes.

4.2 Review of New Zealand Studies

In comparison to the international body of research that has investigated the effects of plantation forest management practices on soil carbon stocks, relatively few studies have yet been undertaken in New Zealand to specifically address this issue. However, some measure of soil carbon concentration or organic matter content, usually only in the topsoil, has been included in several studies of forest soil nutrition under different management regimes (e.g., Smith et al. 1994, 2000; Watt et al. 2005). Recently, research funded by the Ministry for the Environment has been undertaken by SCION to specifically investigate the impacts of some selected forest management practices on soil carbon stocks under certain site conditions (e.g., Oliver et al. 2004; Jones 2007, 2008). The findings of relevant New Zealand studies are summarised below under separate headings for each of the five forest management groups defined above. Further details may be found in Appendix 1 to this chapter.

4.2.1 Silvicultural practices

Several studies have assessed the effects of tree stocking rates on soil carbon stocks in exotic plantation forest and agroforestry sites in New Zealand. One study (Davis et al. 2007) has recently investigated the effects of tree stocking in 5- and 10-year-old, first rotation, *Pinus nigra* stands at the Balmoral site in the upper Waitaki catchment of the South Island. Davis et al. (2007) found that, after 10 years, the stocking rates investigated (250, 500, and 750 stems ha⁻¹) had no effect on mineral soil carbon stocks to 30 cm depth under *P. nigra* at this sub-humid, low-productivity site.

Two earlier studies (Perrott et al. 1999; Saggar et al. 2001) investigated the effects of *Pinus radiata* stocking rates (50, 100, 200, and 400 stems ha⁻¹) in an agroforestry system at Tikitere near Rotorua. Perrott et al. (1999) showed that the carbon stock (and concentration) in the top 7.5 cm of mineral soil generally decreased with increasing stocking rate from about 40 t C ha⁻¹ under 50 stems ha⁻¹ to 30 t C ha⁻¹ under 200 stems ha⁻¹ (i.e. a loss of 10 t C ha⁻¹). However, soil carbon stocks increased slightly (to about 32 t C ha⁻¹) under a stocking of 400 stems ha⁻¹. Subsequent sampling at the same site indicated a reduction in soil carbon concentration of about 1 % in the top 3.5 cm of mineral soil, and smaller reductions with depth, due to an increase in stocking rate from 0 to 200 stems ha⁻¹.

Saggar et al. (2001) examined soil carbon concentrations in mineral soil at two depth increments (0–10 and 10–20 cm) and C stocks in forest floor materials at the Tikitere site. Consistent with the findings of Perrott et al. (1999), Saggar et al. (2001) showed that C concentrations in the top 10 cm of mineral soil were lowest under a stocking of 200 stems ha⁻¹. In contrast to the reported effect on mineral soil carbon concentration, stocks of C in the forest floor (the organic horizon overlying the mineral soil) were found to have increased with increasing stocking rate from 3.7 t C ha⁻¹ under 50 stems ha⁻¹ to 12 t C ha⁻¹ under 200 stems ha⁻¹ and then decreased slightly to 9 t C ha⁻¹ under 400 stems ha⁻¹. Saggar et al. (2001) also showed that stocking rate had no effect on the C concentration in the 10–20 cm depth range, indicating that the effects of stocking rate are most likely restricted to the top 10 cm of mineral soil.

More recently, Scott et al. (2006b) also studied C stocks in forest floor materials (L and FH horizons) and mineral soils (0–10, 10–20, and 20–50 cm depth increments) under different stocking densities (100 and 400 stems ha⁻¹) at the Tikitere agroforestry site. They found that

C stocks in the FH horizon under 400 stems ha^{-1} ($\sim 10 \text{ t C ha}^{-1}$) were almost double those in the FH horizon under 100 stems ha^{-1} (5.7 t C ha^{-1}). However, no significant differences were observed between mineral soil carbon stocks (at any depth) under stocking rates of 100 stems ha^{-1} and 400 stems ha^{-1} . It is possible that if mineral soil carbon stocks under stocking rates of 50 stems ha^{-1} and 200 stems ha^{-1} had been measured and compared, some significant differences may have been observed.

The findings of the above studies, including those of Scott et al. (2006b) and Davis et al. (2007), suggest that soil carbon stocks may only be affected by differences in tree stocking rates at densities up to 200 stems ha^{-1} . However, further studies that measure soil carbon stocks to 30 cm depth and cover a greater range of soil and climatic conditions—rather than just the two sites studied to date—are probably required to confirm this.

There have been no published studies that have comprehensively assessed the impacts of species selection or thinning and pruning regimes on soil carbon stocks in New Zealand, though Carey et al. (1982) found forest floor organic matter levels generally increased with increased stocking. Although not directly assessed by Carey et al. (1982), a corollary of their finding regarding stocking is that increased thinning intensity may decrease organic matter stocks on the forest floor, and thus reduce inputs to the mineral soil. Thinning and pruning regimes currently vary from forest to forest, depending on the intended end-use of the logs, and could potentially lead to differences in organic matter inputs to the mineral soil that co-vary with site and climatic conditions.

Watt et al. (2008) reported that there were no significant differences in C concentrations in the top 10 cm of mineral soil measured under *P. radiata* and *Cupressus lusitanica* across 31 experimental sites in plantation forests throughout New Zealand. However, this was measured after only 4 years growth. Another study is under way on Banks Peninsula, Canterbury, which will provide data on the effect of three species on soil carbon (see Appendix 1, Dataset 12).

In an unpublished litter decomposition study, Baker (1983) measured soil bulk density and organic matter levels (using loss-on-ignition) in the top 5 cm of mineral soil under adjacent plots in 13-year-old *P. radiata* and *Eucalyptus regnans* stands at Whakamaru in the central North Island. The unpublished data from this study could be used to estimate species differences in soil carbon stocks between *P. radiata* and *E. regnans* in the top 5 cm of mineral soil. Similarly, Jurgensen et al. (1986) measured organic matter levels and bulk density in mineral soils (0–20 cm and 10–40 cm depth ranges) under paired plots of an age series of *P. radiata* and *E. regnans* in the central North Island which could be used to estimate soil carbon stocks, assuming an average value of organic C lost during ignition. A limitation of both of these studies is that they are pseudo-replicated. Therefore soil carbon under adjacent plots may or may not have been identical at the time the sites were planted. The lack of research into the potential effects of species and thinning and pruning practices on soil carbon across New Zealand represents a gap in our knowledge that may become more important with time as attention turns to the use of alternative species to *P. radiata* for plantation forestry.

A New Zealand study by Beets et al. (2002) tracked the change in soil carbon that occurred after afforestation over time through to the end of the first rotation. Stocks of C in the top 10 cm of mineral soil were periodically measured at the Puruki experimental site and it was

found that stocks had decreased by about 4 t C ha⁻¹ over the first rotation. Further work is required to determine the magnitude of changes through the second and subsequent rotations.

4.2.2 Forest harvesting

The impacts of two contrasting forest harvesting methods (hauler- and ground-based) on soil carbon concentrations and stocks were investigated at the Puruki experimental catchment, central North Island, by Oliver et al. (2004). They estimated mean soil carbon stocks in the top 10 cm of mineral soil before and after harvesting and reported that hauler-based harvesting—involving the semi-aerial conveyance of felled logs, using cables, towards an elevated collection point—did not significantly alter soil carbon stocks. In contrast to this result, ground-based harvesting—involving the dragging of felled logs across the ground surface using tracked machinery—was shown to have resulted in a significant 5 t C ha⁻¹ reduction in soil carbon stocks in the 0–10 cm depth range. In one sub-catchment, soil carbon stocks were measured by two different methods (tube/auger sampling and power auger sampling with different bulk density estimates) in 6 plots to a depth of 100 cm before and after ground-based harvesting. In this instance, no significant differences due to harvesting were found. In the same sub-catchment, Oliver et al. (2004) also examined the effects of ground-based harvesting on soil carbon concentrations to 200 cm depth. No significant differences between soil carbon concentrations before (measured once only) and after harvesting were detected in any depth range within the studied sub-catchment although trends in the soil carbon concentration data suggested a possible decrease in the 0–20 cm depth range and a possible increase in the 20–50 cm and 50–100 cm depth ranges. These trends probably indicate that some mixing of topsoil and subsoil had occurred due to ground-based harvesting.

In a study of the effects of hauler-based forest harvesting on soil properties and the performance of several soil spatial prediction techniques at a *P. radiata* site north of Auckland (southern Mahurangi Forest) predominantly on heavy clay soils, Jones (2004) found a significant increase of around 1.3 % in the C concentration of the top 10 cm of mineral soil about two years after harvesting. The increase was thought to be due mainly to the growth of an almost complete ground-cover of grasses and weeds across the harvested and replanted area. The decomposition of harvest residues that were retained on the site may also have contributed. Bulk density was measured for each sample point (using a small core method) which allowed for the subsequent calculation of soil carbon stock change due to harvesting at this site. The post-harvested plot had mean soil carbon stocks in the top 10 cm of about 45 t C ha⁻¹ whereas the adjacent pre-harvested plot had a mean stock of about 35 t C ha⁻¹. Therefore, hauler-based harvesting (which caused some soil compaction as evidenced by a significant reduction in mean macroporosity) followed by weed and grass growth and residue decomposition over a 2-year period led to a ~ 10 t C ha⁻¹ increase in the soil carbon stock of the 0–10 cm depth range. However, the observed increase in C stock is probably more due to the retention of harvest residues on-site and the lack of any weed control rather than the practice of hauler-based harvesting itself which probably only contributed by increasing the bulk density.

Forest harvesting activities can, under certain soil texture and moisture conditions, result in soil compaction (Simcock et al. 2006). Watt et al. (2005, 2008) examined the effect of harvesting-related soil compaction on soil carbon concentration in the top 10 cm of mineral soil at 31 forest sites across New Zealand. They found that disturbed (compacted) plots had significantly lower soil carbon concentrations than undisturbed plots before replanting, but

the lower concentrations, together with significantly higher bulk densities, led to increasing soil carbon stocks. Harvesting-related compaction resulted in an increase of soil carbon stocks from 57 to 58.9 t C ha⁻¹ in the 0–10 cm mineral soil depth range (statistical significance not determined). This can only be an apparent increase arising from the increased density of the upper soil layer—if stocks were measured on an equivalent-mass basis (i.e. stocks corrected for the change in bulk density) the soil carbon mass would in fact decrease as the soil carbon concentration declined in this case. Nevertheless, when considering C stocks within fixed mineral soil depth ranges (e.g., 0–10 or 0–30 cm), an apparent soil carbon stock change will result (either in-part or in-full) from a management-induced change in bulk density. In carbon accounting terms, this is a real and valid effect of the management practice, but it is questionable whether it means the same in terms of impacts on atmospheric CO₂ concentrations.

No other published studies have examined the effects of different harvesting methods on soil carbon stocks in New Zealand. The findings of Oliver et al. (2004) and Jones (2004) suggest that the effects might differ considerably between different harvesting methods with those that physically disturb the soil profile potentially causing the loss or displacement of soil carbon and those that, under certain soil or site conditions, do not physically disturb the soil potentially maintaining or even increasing soil carbon stocks. However, these findings are limited to only two sets of soil, climatic, and other site conditions that exist at Puruki and Mahurangi. More work is required to establish whether the effects observed by Oliver et al. (2004) and Jones (2004) are consistent across a wider range of site conditions in New Zealand.

4.2.3 Residue management

Jones et al. (in press) have studied the effects of some common harvest residue management practices (treatments) on C stocks in forest floor materials and mineral soils to 30 cm depth at Tarawera Forest in the Bay of Plenty. Treatments investigated were residue retention (stem-only harvesting), residue removal (whole-tree harvesting), and residue plus forest floor removal (forest floor disturbance). They found that 16–17 years after application of the residue management treatments, the removal of harvest residues plus forest floor material resulted in a significant reduction of C stocks on the forest floor, 0–10 cm total (> 2 mm + < 2 mm fractions) mineral soil, and total soil (0–30 total mineral soil + forest floor) pools in comparison to the retention of harvest residues and forest floor materials. The C stock in the total soil pool was reduced by about 10 t C ha⁻¹ (a reduction of ~ 4 t C ha⁻¹ in the forest floor pool and a reduction of ~ 5 t C ha⁻¹ in the 0–10 cm mineral soil pool) due to harvest residue plus forest floor removal. It should be noted that significant treatment effects on mineral soil carbon stocks were only detected after the C stocks contained in the mineral soil coarse (> 2 mm) fraction were summed with those in the fine (< 2 mm) fraction (Jones et al. 2008). At the Tarawera site the coarse fraction of the mineral soil was found to contain substantial C stocks (~ 5 t C ha⁻¹ average across treatments).

Several earlier studies have also looked at harvest residue management effects on soil carbon or organic matter stocks or concentrations (Ballard & Will 1981; Smith et al. 1994, 2000). Ballard and Will (1981) found that the constant removal of harvest residues plus forest floor materials over a 16-year period following harvesting and re-planting at a pumice soil site in Kaingaroa Forest resulted in the significant decrease in C concentrations in the 5–10 and 10–20 cm mineral soil depth ranges (equivalent to a change in stocks of 2–3 t C ha⁻¹). However, no change in C concentrations in the 0–5 and 20–40 cm depth ranges were observed.

Although soil carbon stocks were not reported, Ballard and Will (1981) measured soil bulk density for the corresponding depth ranges which allowed for the calculation of stocks (given above).

On the sandy soils of Woodhill Forest, located northwest of Auckland, Smith et al. (1994) investigated the effects of four harvest residue treatments 5 years after treatment application, on organic matter (rather than C) stocks in the forest floor and mineral soil to 90 cm depth. The treatments were similar to those studied by Jones et al. (in press) with the addition of a 'double slash' treatment which involved the emplacement of harvest residues collected from the residue removal treatment within plots with a normal quantity of residues already retained. Smith et al. (1994) found large differences in the organic matter stocks of the forest floor materials under the different treatments—the largest stock (82 t ha^{-1}) under the 'double slash' treatment, the smallest stock (2.4 t ha^{-1}) under the residue plus forest floor removal treatment. However, they did not find any significant treatment effects on the organic matter stocks of the mineral soil to 90 cm depth. It is possible that, only 5 years after treatment application, there had not been sufficient time for the differences in organic matter stocks of the forest floors to have resulted in changes to mineral soil organic matter stocks.

More recently, Smith et al. (2000) examined the effects of the three main harvest residue management treatments, previously described, on the C concentration in forest floor materials (unadjusted for loss-on-ignition and therefore including mineral soil contamination) 5 years after treatment application at three forest sites with contrasting soil and climatic conditions (Woodhill, Tarawera, and Kinleith Forests). They observed differences between treatments with the unadjusted C concentration of the forest floor always greatest under the residue plus forest floor removal treatment—presumably due to the generally less decomposed nature of those materials—and generally least under the residue removal treatment. However, comparisons of the forest floor C concentrations between sites and probably even within sites are not valid because they are not adjusted for loss-on-ignition and therefore do not allow for variable soil contamination.

Further information is available for these sites, and one additional site (Golden Downs), from a recent PhD study that examined the residue management effects on C stocks in the FH layer 8–16 years after the treatments were applied (Smaill et al. 2008a). They found that the FH layer in the residue removal and residue plus forest floor removal treatments had 4.4 and 8.4 t C ha⁻¹ less, respectively, than where residues were retained.

In considering the findings of the above experimental studies, it is apparent that much of the work has been focused on soils formed in very sandy (Woodhill Forest) or relatively coarse volcanic parent materials (Tarawera, Kinleith, or Kaingaroa Forests) under carefully applied, experimental management conditions. Further work is therefore needed to establish the effects of harvest residue management under a more complete range of soil, climatic and operational management conditions in New Zealand. Nevertheless, there is some strong evidence to suggest that the removal of harvest residues plus forest floor materials can result in a significant reduction of soil carbon stocks.

Harvest residue management may also involve the burning of residues before re-planting. Robertson (1998) determined the C stocks that are lost as the result of the prescribed burning of the harvest residues from two species (*P. radiata* and *Pinus contorta*) at five sites across New Zealand (Glenbervie, Mawhera, Topuni—two sites, and Kaingaroa Forests). The *P. contorta* site was located within Kaingaroa Forest. The effect of burning on soil carbon

stocks was also investigated at four of the five sites. Stocks of C in *P. radiata* residues were found to have been reduced by about 27 t C ha⁻¹ (representing a 62% reduction) due to burning whereas stocks of C in *P. contorta* were found to have been reduced by around 33 t C ha⁻¹ (representing a 48 % reduction). However, little or no effect on C stocks in the top 10 cm of mineral soil was observed (Robertson, 1998). The impacts of mechanical manipulations of harvest residues (e.g., windrowing) on soil carbon stocks have not been comprehensively assessed in New Zealand to date although one early study by Ballard (1978) partially investigated the effects of windrowing (discussed below).

4.2.4 Mechanical site preparation

Few studies have investigated the effects of different mechanical site preparation practices on soil carbon stocks in New Zealand. Ballard (1978) examined the effects of windrowing and skid site formation (i.e. surface soil scarification) on soil properties and the productivity of a 7-year-old *P. radiata* stand growing on a yellow-brown pumice soil in Kaingaroa Forest in the central North Island. He found significantly lower total C concentrations in 0–5 cm, 5–10 cm, and 10–20 cm mineral soil depth ranges under skid sites, but not under inter-windrow areas, compared with undisturbed areas. Soil under windrows themselves could not be sampled in the same way because of irregular mixing of harvest residues with the mineral soil. Using total N concentration as an indicator of depth of profile development, Ballard (1978) calculated that 2.5 cm and 25.7 cm of soil had been removed from inter-windrow and skid site areas respectively by the windrowing operation, which explains why soil carbon concentration from 20 to 30 cm depth on skid sites was not significantly different to the undisturbed profile. Although soil bulk density was not given, Ballard (1978) estimated that about 700 kg of N would be removed per ha in 2.5 cm of topsoil and, with a C:N ratio of 18.5 (in the top 5 cm), it appears that almost 13 t C ha⁻¹ of WHAT? would also be removed in the same 2.5 cm of topsoil.

Jones (2007) recently studied the impacts of two different forest soil cultivation techniques (spot-mounding and line-ripping) at two different sites with contrasting soil conditions (Rotoehu in the Bay of Plenty and Lochinver in the central North Island). At both sites, soil carbon stocks in disturbance features to 30 cm depth were compared with those in adjacent undisturbed soil. The slopes of spot- and rip-mounds were adjusted for in the calculation of C stocks. Net, area-adjusted, soil carbon stock changes were determined for each site which took into account the fact that forest soil cultivation disturbs only about 30 % of the total land area subjected to this management practice (Jones 2007). It was found that spot-mounding resulted in a significant net, area-adjusted, reduction in C stocks of about 4 t C ha⁻¹ in the top 30 cm of mineral soil 15 months after cultivation at the relatively fertile Rotoehu site, whereas the line-ripping of a relatively infertile site (Lochinver) was found to have had no significant effect on the C stocks in 0–30 cm depth range 38 months after cultivation.

At Lochinver a significant reduction of more than 2 t C ha⁻¹ in the C stocks of the top 10 cm of mineral soil was offset by a 1 t C ha⁻¹ gain in the 20–30 cm depth range (Jones 2007). No other relevant studies have been undertaken in New Zealand to date. Clearly, there is a need to determine the impacts of forest soil cultivation and other mechanical site preparation practices on soil carbon stocks across a much wider range of soil and climatic conditions in New Zealand. Direct comparisons of alternative soil cultivation techniques on similar soils are also required to establish which are best in terms of conserving soil carbon stocks while continuing to function effectively as a seedling-growth promoting activity.

4.2.5 Site improvement

Few studies have examined the effects of site improvement practices (fertiliser application and weed control) on soil carbon stocks in New Zealand. However, some studies of forest soil nutrition have measured changes in soil carbon concentrations and organic matter stocks (Baker et al. 1986; Smith et al., 2000) and one study has investigated the effect of understorey (grass) removal in an agroforestry system (Chang et al. 2002).

Smith et al. (1994) reported that 5 years after harvesting a *P. radiata* forest (Woodhill Forest), there were no significant differences in the organic matter stocks on the forest floor or in the sandy mineral soil (to 90 cm depth) between fertilised (receiving 200 kg N ha⁻¹ year⁻¹) and unfertilised plots. However, they stated that trends in the data suggested that, over a greater period of time, the application of the N fertiliser may lead to increased stocks of organic matter in the mineral soil. In a subsequent study that measured C concentrations in forest floor materials at three trial sites across the North Island (Woodhill, Tarawera, and Kinleith Forests), Smith et al. (2000) found that the application of N, P, and other fertilisers had had little impact on forest floor C concentrations 5 years after harvesting.

Since both of the above studies examined fertiliser effects after only a relatively short time (5 years) it is possible that significant changes in soil carbon stocks may have subsequently occurred at the sites investigated. For instance, Baker et al. (1986) reported a significant (more than two-fold) increase in the concentration of soil organic C in the 0–5 cm depth range after the application of mixed fertiliser (including 960 kg N ha⁻¹ over 10 years) to the sandy soils in Woodhill Forest. Much more recently, Watt et al. (2008) have found that mixed fertiliser (N, P, K, Mg, S, and Ca) application led to a significant increase in topsoil (0–10 cm) C concentration measured after harvesting of 4-year-old, densely-stocked plots across 31 experimental sites in plantation forests throughout New Zealand. The influence of fertiliser application on C stocks was not reported in the paper, but could be determined as soil bulk density was measured. However, further work still needs to be undertaken to consistently and systematically quantify the changes in soil carbon stocks (rather than just C concentrations) to 30 cm depth over longer time periods following forest fertilisation (as applied in management operations) under a comprehensive range of site conditions in New Zealand.

Understorey management (weed control) is another aspect of site improvement that may potentially alter soil carbon stocks in exotic plantation forests. These effects have not been widely and comprehensively studied in New Zealand. Chang et al. (2002) considered the effect of the presence and absence of a pasture understorey at a *P. radiata* agroforestry site on the Canterbury plains.

Their results showed that the absence of the ryegrass pasture (bare ground) lead to a significantly lower C concentration in the top 10 cm of mineral soil compared to the soil with a pasture understorey. However, no effect was observed in the 10–20 cm depth range. This finding (and that of Jones 2004—see Forest Harvesting above) suggests that the grassy/weedy understorey vegetation that often occurs in plantation forests—particularly before canopy closure—may, at least under certain circumstances, help maintain or increase surface soil carbon stocks during stand establishment. Much more work is required, however, to fully investigate and test this hypothesis under a range of site and understorey conditions in New Zealand.

4.3 Review of New Zealand Datasets

In total, we identified and described 14 readily accessible datasets that contain some information on the impacts of various forest management practices on soil carbon stocks or concentrations in exotic plantation (refer to Appendix 1 for full dataset descriptions). All the identified datasets are experimental, with the exception of the Kyoto Forest Inventory dataset, meaning that the results they produced may be quite specific to the carefully designed treatments and site conditions under which they were applied. The treatments may or may not be particularly representative of the management practices that are routinely applied in general forest operations depending on the objectives of the experimental design. Moreover, the effects of operationally applied management practices are likely to be more variable than those measured under carefully controlled experimental conditions. It should be noted that most of the datasets were developed for purposes other than the systematic accounting of soil carbon stocks and stock changes. In fact, to date there has been no systematic inventory of soil carbon stocks under exotic plantations—unlike that undertaken as part of the implementation of the indigenous carbon monitoring system (Davis et al. 2004).

The main overall weaknesses of the available soil carbon datasets relating to forest management practices in terms of their usefulness for C accounting include the ad hoc assessment of the effects of selected management practices under specific (or a limited range of) soil and site conditions, and inconsistent sampling and C measurement methods and protocols (e.g., inconsistent sampling depths, different sampling intensities, and lack of bulk density measurement).

Each individual dataset has its own particular strengths with some having reasonable geographic spread (e.g., LTSP II (Long Term Site Productivity II) and others having C stock data to 30 cm depth based on intensive and systematic sampling (e.g., Rotoehu and Lochinver studies).

4.3.1 Silvicultural practices

There are a number of datasets available in New Zealand that, at least in part, could provide some information on the effects of tree stocking rates (i.e. Balmoral-Waitaki; Dataset 11 and Tikitere; Dataset 9), species selection (LTSP II; Dataset 7, LTSP III; Dataset 8, and Orton Bradley; Dataset 12) and age class effects (Kyoto Forest Inventory; Dataset 10—under development). Findings from some of these datasets have been published or reported, and have been discussed above (e.g., Perrott et al. 1999; Saggar et al. 2001; Scott et al. 2006b; Davis et al. 2007; Watt et al. 2008). Of the various silvicultural practices included in this management group, stocking rate has been the most studied (in terms of publications) but the datasets this work was based on cover only two sites. The lack of geographic spread in those data creates uncertainty as to whether the effects of stocking rate that have been identified at Tikitere, for example, occur under different soil and site conditions in New Zealand. The data relating to the effects of species selection are limited because relatively few species have been compared to date (*P. radiata* compared with *C. lusitanica* or *Cupressus macrocarpa* and *E. nitens*). However, there is potential for further comparisons to be made between *P. radiata*, Douglas-fir (*Pseudotsuga*) and cypress (*Cupressaceae*) species using the LTSP III trial sites in the future.

4.3.2 Forest harvesting

Two studies have been undertaken on the effects of forest harvesting on soil carbon. Both studies are restricted to single sites in Northland (Mahurangi) and the central North Island (Puruki), limiting the ability to apply the results elsewhere. Results from both studies have been reported and are discussed above (i.e. Oliver et al. 2004; Jones 2004). The Puruki study (dataset 5) compared hauler and ground-based logging impacts by measuring soil carbon before and after harvest. Although the treatment comparisons were made in separate sub-catchments, the approach of measuring before and after harvesting is robust and indicates that harvest methods differ in their impacts on soil carbon. However, the data below 10 cm are constrained by a relatively low number of plots and the ability to detect differences of less than 30 % between treatments is therefore limited. The Mahurangi study (dataset 4) is limited by the fact that it compared soil carbon in adjacent, non-replicated pre- and post-harvested plots, meaning that differences cannot be unequivocally attributed to a harvesting or other site management effect. However, a similar effect was found by Chang et al. (2002), as noted previously.

A further study (LTSP II trial series, Dataset 7) examined the effect of harvesting disturbance, mainly through compaction, on soil carbon 4 years after planting. The trial series was located at 31 sites nationwide encompassing all soil orders important for forestry; thus the results are widely applicable. A constraint of this dataset is that it is not possible to determine if the effect of disturbance/compaction persists for any length of time because the trials were harvested at age four. Results have been presented as a mean effect across all soil orders (Watt et al. 2008) and so further data analysis would be required to determine compaction effects for individual soil orders.

4.3.3 Residue management

Residue management effects on soil carbon have been examined in the LTSP I trial series (dataset 6) which was established between 1986 and 1994 at six sites, three in the North Island and three in the South Island. The sites were located on Recent, Pumice, Pallic, and Brown soils, thus the series has good coverage of important forest soil orders. Further strengths are that all six sites were well replicated and forest floor and mineral soil (0–20 cm) C pools were determined at establishment. Limitations include the fact that baseline data was not collected for all plots at all sites and only site-averaged data were collected at two of the sites. Since establishment, two sites have been converted from forest to dairy pastures. One of the sites (Tarawera; Dataset 1) has been studied 16–17 years after application of the residue management treatments and results are in publication (Jones et al., in press; discussed above). Four sites were studied as part of a PhD project that included measurement of treatment effects on FH layer C contents at mid-rotation age (Smaill et al. 2008a; discussed above). At the now deforested sites, soil samples were collected immediately before deforestation but are yet to be fully analysed and reported (in progress). Therefore, more data on residue management impacts on mineral soil carbon, in addition to effects on forest-floor carbon, are potentially available, at the mid-rotation stage for three of the six sites. Further sampling and analysis is required to determine residue management effects at the remaining sites.

4.3.4 Mechanical site preparation

Two studies are relevant here—the Rotohehu spot-mounding study (dataset 2) and the Lochinver line-ripping study (dataset 3). The results of both are discussed above (Jones 2007). These studies are well replicated and strengths of both are that the treatment effects were calculated on an area basis. Although both studies are limited to single sites, the trends of the treatment effects found may be similar at other sites. However, the size of soil carbon stock changes (magnitude of treatment effects) is likely to be dependent on initial soil carbon stocks at a given site (i.e. site dependant). Therefore, the use of proportional rather than absolute treatment effects may be the best way to apply the data for C accounting purposes at this stage. More work is required to establish the magnitude of soil-carbon stock changes under different soil and site conditions.

4.3.5 Site improvement

The LTSP II trial series (dataset 7) examined the effect of fertiliser (a mixture of all nutrients likely to be limiting growth) on soil carbon four years after planting (Watt et al. 2008; see above). The trial series is located at 31 sites nationwide on all soil orders important for forestry and so the results for soil carbon accounting purposes are widely applicable. As noted above, Watt et al. (2008) found that fertiliser application led to a significant increase in topsoil (0–10 cm) C concentration. Changes in C stocks were not reported, but could be determined as soil bulk density was measured. A constraint of the study is that it is not possible to determine if the effect of fertiliser addition on soil carbon persists beyond 4 years—the age at which the trials were harvested. A further constraint arises from the fact that the fertiliser applied was not a standard application as commonly applied in forest operations. An additional dataset (Dataset 14), developed primarily to assess the growth response of the mid-rotation application of N and P fertilisers, contains some soil carbon data and so could be analysed in the future to further examine the effects of fertiliser addition on soil carbon stocks.

The LTSP II trial series—in addition to small, highly stocked plots which were harvested at age four—has companion 20 × 20 m permanent sample plots located at each of the 31 sites, from which initial soil carbon measurements have been taken. In these plots, herbicide control of weeds has been applied to half the plot but not to the other half; thus presenting the opportunity to assess the effect of weed control on soil carbon stocks, and how any effects might vary with soil order.

Another ongoing trial series also contains treatments in which the effect of herbicide weed control on soil carbon stocks could be determined (dataset 13). This is a well-replicated trial series located at four sites (two in each of the North and South Islands) with contrasting rainfall and boron availability (the main focus is on boron availability). Soil samples have been collected but have not yet been analysed for soil carbon concentrations. If the samples were to be analysed for total C in the future and soil bulk density measured at the sites then this might provide a useful dataset to help establish the effect of weed control on soil carbon stocks in New Zealand.

4.4 Review of Key International Information

A review of some of the key international literature was undertaken by Jones (2005). The key findings of that review were:

- Forest management activities that resulted in the preservation of harvest residues, forest floor materials, topsoils, and weeds on site or that mixed organic materials into the mineral soil tended to increase soil carbon stocks at least in the short term.
- Harvesting and other soil disturbance-inducing activities were generally shown to increase the variability in soil carbon stocks whereas prescribed burning tended to reduce variability.
- The impacts of forest management on soil carbon stocks are likely to be very site specific and may differ between forest regions within a country.
- The spatial variability in soil carbon is a key consideration in the development of an assessment and monitoring system. It needs to be accounted for and necessitates the use of adequate replication and sample sizes.

4.4.1 Silvicultural practices

Jandl et al. (2007) have reviewed the effect of thinning on soil carbon stocks. Thinning changes the site microclimate and may result in soils becoming warmer and perhaps moister because of decreased evapotranspiration which could, in turn, lead to increased organic matter decomposition. Forest floor C stocks have been shown to decrease with increased thinning intensity in field studies in a number of countries (including New Zealand). However, these effects are probably only transitory.

There is little information on the effect of thinning on mineral soil carbon stocks (Jandl et al. 2007). An experiment in Austria with Norway spruce showed thinning reduced mineral soil carbon. However, stockings were much higher than those used in New Zealand (5700 reduced to 2100 stems/ha). Another study in Finland showed no effects of thinning on ecosystem C because reduced C sequestration by trees was compensated for by enhanced growth of understorey vegetation (Jandl et al. 2007). These studies are, in general, not relevant to New Zealand conditions.

4.4.2 Forest harvesting

Harvesting removes biomass, disturbs the soil, and changes the site microclimate, all of which can affect soil carbon stocks. In a review that included a meta-analysis (73 observations from 26 publications), Johnson and Curtis (2001) found that, on average, forest harvesting had little effect on soil carbon in the A horizon or the whole soil (forest floor data not included in the analysis). However, stem-only harvesting caused increases (18%) in soil carbon in the A horizon while whole-tree harvesting caused decreases (6%). The increase was restricted to conifer forests and may have resulted from incorporation of residues in the soil but could also be considered more an effect of harvest residue management—stem-only harvest retains residues while whole-tree harvest removes them—rather than forest harvesting disturbance *per se* (see residue management section below). Subsequent to that review, Johnson et al. (2002) reported that, on balance, there were no long-term harvesting impacts on soil carbon stocks after comparing three different harvesting/residue management types (stem-only, whole-tree, and whole-tree with stumps removed) at four sites in the south-eastern USA. Similarly, a review of harvesting techniques by Jones (2005) found that various studies either showed no change, small changes, or short-term changes in soil carbon

after harvesting. Larger soil carbon losses following harvesting have been reported by some researchers (see review by Jandl et al. 2007), but these studies are restricted to conifer forests at high northern latitudes. As the impact of harvesting on soil carbon may depend on the method of harvest (e.g., hauler or ground-based), the overall results of the meta-analysis of Johnson and Curtis (2001) are not immediately relevant to the New Zealand situation. Further stratification of the studies by harvest method (to select studies that used harvest methods similar to those used in New Zealand) would be needed before their results could be applied here.

4.4.3 Residue management

As noted above, the meta-analysis of Johnson and Curtis (2001) indicated that removal of harvest residues (whole-tree harvesting) caused a reduction in A horizon soil carbon stocks, whereas retention of residues on site (stem-only harvesting) caused an increase in A horizon soil carbon in conifer forests. The shortcomings of this study in terms of applying the results to the New Zealand situation have been noted above. Again, further stratification of the individual studies by harvest method would be required for the results to be usefully applied here. Furthermore, residue management practices compared in international studies may not always correspond to standard practices applied in New Zealand.

Substantial amounts of the C (in excess of 30 to 50 t C ha⁻¹) contained in harvest residues can be lost to the atmosphere if it is burnt (Mendham et al. 2003). Consistent with the lower end of the estimate range, Robertson (1998) reported a 27 t C ha⁻¹ loss of C to the atmosphere from burning *P. radiata* residues and 33 t C ha⁻¹ loss from burning *P. contorta* residues in New Zealand. The study of Robertson (1998) was based on four sites for *P. radiata* and so should adequately represent New Zealand conditions and not require supporting international data. The meta-analysis of Johnson and Curtis (2001) showed no overall effects of fire on mineral soil carbon in either the A horizon or whole soil. Similarly, Robertson (1998) found no evidence of carbon loss or gain in the top 10 cm of the mineral soil under New Zealand conditions. Some international studies have shown quite large losses in mineral soil carbon after fire (Black & Harden 1995; Antos et al. 2003) but these were undertaken in old-growth Douglas fir and mixed conifer forests in the north-western USA and may not be comparable to plantation grown radiata in New Zealand. The use of prescribed burning of harvest residues is probably not as common in New Zealand as it is internationally (particularly in the USA).

4.4.4 Mechanical site preparation

From their review of forest management effects on soil carbon, Jandl et al. (2007) concluded that site preparation generally causes a net loss of soil carbon. However, Ryan et al. (1992) reported that mounding and rutting (involving the mixing of organic and mineral soil materials) significantly increased the pool of soil total C in comparison to undisturbed soil. Soil bedding in loblolly pine stands was also found to result in a short-term increase in soil total C in the 0–15 cm depth range (Carter et al. 2002). These and other studies (Schmidt et al. 1996; Trettin et al. 1996) indicate that the effects of site preparation on soil carbon stocks are likely to be highly dependent on the method used as well as initial soil and site conditions. Because of this, studies elsewhere are unlikely to be relevant to New Zealand conditions or useful for addressing gaps in the limited New Zealand data.

4.4.5 Site improvement

A meta analysis of the effect of fertiliser and N-fixing species on soil carbon showed that both fertiliser and N-fixers generally increased A-horizon and whole soil carbon concentration, with no significant differences between the two (Johnson & Curtis 2001). However, results varied widely across the different studies. The meta-analysis included 48 observations from 16 publications. In contrast, Homann et al. (2001) found from a study of 13 second-growth Pacific Northwest Douglas fir stands that urea fertilization had no effect on C mass or concentration in the forest floor or mineral soil layers 10 years after application. However, variation precluded detection of differences of less than 15%. Also Moscatelli et al. (2008) found no increase in soil carbon after fertilising poplar stands in Italy, and Shan et al. (2001) found no effect of fertiliser on soil carbon in slash pine plantations in the southern USA. Thus, fertiliser can have widely varying results on C storage in forest soils. Effects are site specific and no general recommendations on a regional level can be made (Jandl et al. 2007), indicating generalised results from international studies should not be applied to New Zealand.

The use of herbicides to control weeds after harvesting and site preparation treatments has been shown to reduce total soil C in a number of studies (see review of Jones 2005). These data mainly relate to the effects of herbicide in pines, and could be relevant for New Zealand conditions. However, there is potentially data available on the effect of herbicide on soil carbon for a wide range of New Zealand sites from the LTSP II study (dataset 7) and these should be used in preference. As these studies have indicated, the retention of weeds after clear-cut harvesting and site preparation (especially prescribed burning) may assist in maintaining soil organic C stocks (Carlyle 1993).

4.5 Mitigation Opportunities for Forest Soils

Assessment of forest management options

From the above review of New Zealand and international studies, the following forest management options have been identified that could help maintain or increase soil carbon stocks:

- Retention of residues and forest floors on site.
- Retention of a grass or weed cover.
- Avoidance of unnecessary soil disturbance or cultivation.
- Maintenance of soil fertility including fertiliser applications.

4.5.1 Retention of harvest residues and forest floor litter on site

Datasets 1 and 6 provide relevant information in relation this mitigation option. Dataset 1 is an in-depth study of treatment effects (excluding the fertiliser sub-treatments) at the mid-rotation age of one (Tarawera) of the six sites of Dataset 6 (Jones 2005; Jones et al. 2008). These datasets, their strengths and weaknesses and associated publications have been discussed in detail under the 'residue management' heading in the previous sections on 'New Zealand studies' and 'New Zealand datasets' above. Treatments included stem-only harvesting (residues retained), whole-tree harvesting (residues removed) and whole-tree

harvesting plus forest floor removed; sub treatments included plus and minus fertiliser. Four sites were part of a PhD study that included measurement of treatment effects on FH layer and 0–2.5 cm mineral soil organic matter and C contents at mid-rotation age (Smaill et al. 2008a). Two of the six sites have now been converted to dairying, but soil samples were collected (but have not yet been analysed) prior to conversion. Because of geographical spread, coverage of important plantation forest soil groups and sound design, the datasets—supplemented with additional sample collection and analysis—offer a unique opportunity to critically assess long-term residue management (and fertiliser) effects on forest floor and mineral soil carbon stocks in New Zealand. The major limitation to this assessment is a current lack of available funding.

In terms of a mitigation opportunity, the forest floor removal aspect of these studies has some limitation for wider application as forest floors were completely removed off-site. In practice, forest floors are generally disturbed rather than removed completely. An understanding of the impact of residue removal on soil carbon could become important in future if forests are harvested for energy production and whole crowns are harvested. Nevertheless, it is recommended that, wherever possible, forest floor materials and harvest residues be retained on site to help maintain soil carbon stocks as well as the supply of nutrients (Jones et al. 2008). Further research focussed on this issue may be able to identify areas and circumstances where forest floor and harvest residue removal will not necessarily result in soil carbon stock reductions.

4.5.2 Retention of a grass or weed cover

There are a number of indications, from both the New Zealand (Chang et al. 2002; Parfitt et al. 2003; Jones 2004) and international literature (see review of Jones 2005), that retention of ‘weedy’ vegetation through the harvest and re-planting phase contributes to an improvement in soil carbon stocks. Key New Zealand information is contained in Dataset 7—the permanent sample plots associated with the LTSP II trial series. These plots have plus and minus weed control treatments maintained to age four. Soil carbon was measured in soil samples (0–10 cm) taken at plot establishment, and again at age four. This is a particularly strong dataset as it covers 32 sites geographically spread to cover all the major climate and soil domains important for plantation forestry in New Zealand. Although the laboratory analyses have been completed, no reports are available for this dataset as yet. A deficiency with the dataset is that annual assessments have not been made, but a key strength is that there is an opportunity to measure ongoing effects.

A trial series investigating the effect of different boron application rates (dataset 13) is an additional study that could provide data for assessment of ‘weed’ retention effects on soil carbon. This trial has four sites—two in the North Island and two in the South Island. Three of the sites have treatments where weeds have either been controlled or not controlled for several years. These are well-replicated trials where initial soil samples have been collected (but not yet analysed). Reports are available for this trial series, but no soil carbon data have been presented.

Active retention of a grass or weed cover through the harvest phase seems a viable option for retaining soil carbon stocks, or at least mitigating soil carbon losses. Management steps that can be taken include over-sowing of pasture species where ‘weed’ invasion is slow or limited, coupled with fertilisation if necessary, and minimising the herbicide application area around trees to that necessary to achieve good establishment.

4.5.3 Avoidance of unnecessary soil disturbance or cultivation

Soil disturbance occurs during harvesting and when sites are cultivated in preparation for planting. The hauler and ground-based harvesting comparison at Puruki (Oliver et al. 2004, Dataset 5) is a before and after study of the effects of harvesting on soil carbon stocks. The study is limited to one site but results should be applicable to forests covering a large area of the central volcanic plateau of the North Island with similar soils. Although the results indicate there may be mitigation opportunities through use of hauler rather than ground-based logging techniques, further similar studies over a range of sites and soils are required before the results can be applied more widely, and to determine the size of mitigation opportunity. The data of Watt et al. (2008) relating to soil compaction caused by harvesting operations, which were derived from 31 sites located throughout the country, are limited by the depth of sampling (0–10 cm). Reliance on this data set would give an erroneous picture of the effects of soil compaction by machinery during harvesting.

Cultivation generally leads to a reduction in soil carbon stocks, but the effects are highly site specific and dependant on the type of cultivation employed. Only two New Zealand datasets—Dataset 2 (spot mounding) and Dataset 3 (line ripping)—on the effects of cultivation are available. Relevant publications are Jones (2005, 2007). In those studies, spot-mounding significantly reduced soil carbon but line-ripping produced a non-significant reduction. The studies are robust, but limited to single sites and soil condition. Nevertheless, the cultivation treatments are likely to cause similar trends at other sites and information on the proportional change determined from these data could be applied elsewhere to determine potential cultivation effects. As the mitigation opportunity consists of simply not cultivating sites, they would be very easy to apply. However, their use is limited by management constraints. Cultivation techniques like spot-mounding and line-ripping are generally applied to achieve adequate seedling survival and early productivity. The option of not cultivating before seedling establishment there needs careful assessment because gains arising from not cultivating may be outweighed by losses from reduced productivity (and C sequestration in biomass).

4.5.4 Maintenance of soil fertility including fertiliser applications

It is generally believed that fertiliser application, especially N fertiliser, can increase soil carbon stocks. The chief supporting datasets in New Zealand are the LTSP series I and II trials (datasets 6 and 7). The LTSP II series has been completed and results published (Watt et al. 2008). The strength of the study derives from the wide geographic coverage of the field sites which are located across all key environmental domains important for plantation forestry. Deficiencies are that fertiliser was applied as mixtures, and amounts applied depended on site (it was applied to overcome any potential deficiencies), and secondly the short-term nature of the study (despite the name)—effects were determined only up to 4 years after planting. Published results (Watt et al. 2008) indicate the overall effect of fertiliser on soil carbon stocks—additional analysis would be necessary to derive results for individual soil orders or groups.

The LTSP I series is an ongoing field experiment initially established at six sites on important forest soils throughout the county. Relevant strengths and weaknesses of this dataset are described in the ‘Retention of residues and forest floors on site’ section above. Sub-treatments include plus and minus N fertiliser application. Its usefulness is limited by the

total amounts of N applied—these are greater than amounts that are normally applied in forest operations. Effects of N fertiliser on forest floor and the upper mineral soil (0–2.5 cm) were determined at mid-rotation (Smaill et al. 2008b). The shallow depth of sampling of the mineral soil in this study limits the usefulness of results for determining N fertiliser effects on mineral soil carbon stocks.

Additional studies that could be used to provide data on the effect of fertiliser on soil carbon stocks include:

- Mid-rotation N and P trials (dataset 14), where these fertilisers have been added alone and in combination at 5 sites throughout the country. Strengths of this study include good geographical coverage and application of fertiliser at operational rates. Soil sampling at establishment has been done and samples were archived.
- An LTSP III series (dataset 8) site located in Woodhill Forest, where N has been included as a variable. This trial series has also been repeated in Kaingaroa forest central North Island, not with N but as a Mg rates trial onto *P. radiata*.

While the option to increase soil carbon stocks through fertiliser application appears viable, further work is required to establish the likely magnitude of potential increases in soil carbon stocks under different site and soil conditions, and whether such increases are permanent or merely transitory.

4.6 Environmental Co-benefits and Risks

Potential environmental co-benefits and risks associated with different forest management practices include:

Retention of residues and forest floors on site

Residue retention may possibly lead to increased fire risk on some sites (particularly in areas with long, dry summers). Residue retention may also suppress regenerating weed vegetation resulting in a lost opportunity for enhanced soil carbon sequestration via the development of a weedy ground cover. Residue retention has been shown to increase the risk of harm to young *P. radiata* trees in Woodhill Forest in a LTSP series 1 trial where the double slash treatment led to a build-up of thrips which caused needle necrosis and potential growth loss. This infestation required remedial spraying.

Retention of grasses-weed cover

Retention of a vegetation cover during afforestation and through the harvest phase will have co-benefits of reduced risk of soil erosion and stream sedimentation, and reduced flooding potential. On fertile, N-rich sites, retention of a grass or weed (non-leguminous) cover will reduce the potential for N-leaching from catchments into receiving waters (Parfitt et al. 2002).

Avoidance of unnecessary soil disturbance or cultivation

Avoidance of unnecessary soil disturbance or cultivation may reduce the risk of soil erosion and stream sedimentation. However, there is the risk of seedling death, damage, or lack of vigour if cultivation is not performed in areas susceptible to frost, excessive wetness, or where some subsurface impediment exists. Seedling damage may limit future tree growth and potential for C sequestration in above-ground biomass.

Fertiliser application

Use of nitrogen fertiliser to maintain or increase soil carbon stocks may increase the risk of N leaching, especially on sandy soils. This risk may not be acceptable for forests in catchments which drain into receiving waters where maintenance of high quality water is important (e.g., Lake Taupo). On the other hand, appropriate use of N and other fertilisers may not only help improve soil carbon storage but may boost forest productivity, thus increasing C storage within forest biomass.

4.7 Biochar as an Emissions Offset Option

4.7.1 Operational and economic feasibility

Although little information is available on the benefits of applying biochar in forest ecosystems, woody residues are recognised as one of the most important feedstock materials for bio-energy recovery and biochar production (Okimori et al. 2003; Ogawa et al. 2006; FAO 2008).

A recent study showed that woody residues from forests represent the largest, relatively unutilised, biomass resource currently available in New Zealand (Hall & Gifford 2008) and that this could be used for bioenergy or biochar production. Currently, there are about 3.6 Mt of woody residues available from the 1.7 Mha of pine plantation forest, and this is expected to increase to 8.4 Mt by 2026–2030. These residues are deposited either in the forest or on central landings (skid sites). In the forest, (cutover) large trees frequently break during harvesting operations. These broken sections are often too small to be extracted to the landings and are left on-site together with branches from where they are eventually lost by natural decomposition. On skid sites, tree-length stems are cut into logs. Off-cuts from the base, tip, and midsections of stems average about 5% of the extracted volume and usually become waste material. Currently, only about 250 000 t/ year, equivalent to 7% of the total forest harvest residue resource is collected from some skid sites and used to fuel energy plants at wood processing facilities (Hall & Gifford 2008). If these woody residues are left in the forest to decompose, they will release a proportion of their C content as CO₂ and even methane—a greenhouse gas that is 21 times more potent than CO₂—to the atmosphere. It would be preferable to harvest these residues and turn them into biochar rather than to let them all decompose, at least from an emissions-reduction perspective. If woody residues can be economically collected and used for bioenergy or biochar production using pyrolysis technology, it would not only produce environmental benefits, but also generate additional revenue.

4.7.2 Potential impacts on soils and forest ecosystems

Biochar application has been reported to increase soil fertility and plant growth. However, most studies on the potential beneficial effects of biochar application on soil properties and crop responses have been conducted on agricultural soils (e.g., Glaser et al. 2002; Lehmann et al. 2003; Liang et al. 2006; Chan et al. 2007; Lehmann 2007). Very few studies have examined the effects on forest soils (e.g., Berglund et al. 2004). A number of investigations of forest fire-derived biochar indicated that biochar may improve microbial activity in boreal (Zackrisson et al. 1996; DeLuca et al. 2002) and temperate forests (DeLuca et al. 2006). Biochar can contribute a considerable proportion to soil organic C stocks in forests affected by historical fires (Zackrisson et al. 1996; Rumpel et al. 2006).

Nearly all relevant investigations demonstrated that biochar breaks down extremely slowly and persists in the soil for thousands of years (Lehmann et al. 2003; Lehmann 2007). Investigations of the potential effects of biochar on soil productivity and organic C sequestration are overwhelmingly positive (e.g., Lehmann 2007; Chan et al. 2007; Harris & Hill 2007). It is becoming more widely acknowledged that adding large quantities of biochar to soils can help combat global warming by increasing ecosystem C sequestration (Lehmann 2007). Warnock et al. (2007) suggested that biochar application to soil can also have positive effects on the abundance of mycorrhizal fungi. They identified four possible mechanisms by which biochar might influence mycorrhizal fungi abundance. These mechanisms include: (i) alteration of soil physico-chemical properties; (ii) indirect effects on mycorrhizae through effects on other soil microbes; (iii) plant–fungus signalling interference and detoxification of allelochemicals on biochar; and (iv) provision of refugia from fungal grazers. Mycorrhizal fungi are well known for improving plant uptake of nutrients and the relationship between mycorrhizal fungi and charcoal may play an important role in realising the potential of biochar to improve fertility and enhancing C sequestration (Warnock et al. 2007).

However, findings of a recent study by Wardle et al. (2008) suggest that the postulated C sequestration benefits of applying biochar to soils may be overstated. In a field study, wood-derived biochar was prepared and mixed with forest soil humus materials, and left on the soils of three contrasting forest stands in northern Sweden for ten years. It was found that when biochar was mixed into humus, it caused the loss of native soil organic matter due to a significant increase in soil microbial activity (Wardle et al., 2008). In a study on the effects of glucose on microbial decomposition of biochar in soils, Hamer et al. (2004) found that some micro-organisms were able to live with biochar as sole C source. They also found that biochar in soils may enhance the rate of decomposition of labile C compounds. It is likely that biochar application to other soils may accelerate the decomposition of native soil organic C, which could potentially discount the benefits of using biochar to improve soil C sequestration.

4.7.3 Optimising biochar production and application

Ideally, biochar should be produced in such a way that syngas and bio-oil are collected from a pyrolysis facility and utilised for energy recovery (e.g., electricity generation), with biochar used to improve soil productivity and contribute to long-term C sequestration (Lehmann 2007). Although most advancement in pyrolysis research in recent years has been focused on maximising energy recovery in the form of bio-oil with minimal char production (Bridgwater 2005), there has been some research and development in relation to modernising biochar production (e.g., Flash Carbonization technology developed by the University of Hawaii; Grønli et al. 2005). Modern pyrolysis processes can produce bio-oil, biochar, and syngas, with the proportions of each product depending on the temperature and vapour residence time. Bio-oil production requires moderate temperatures and a short residence time, but biochar production requires lower temperature and a longer residence time. There is always a competition between optimising for either maximum energy or maximum biochar production. For example, BEST energies Inc. (www.bestenergies.com) developed a slow pyrolysis system that converts biomass to syngas with up to 35% biochar. Dynamotive employs a fast pyrolysis process that converts biomass to bio-oil, syngas, and biochar, with a yield of 60–75% oil, 15–20% char, and 10–20% gases (www.dynamotive.com).

Although any biomass can be pyrolysed, the process requires biomass to be dried (<10% moisture) and size reduced (<10 mm) before pyrolysis (Hall & Gifford 2008). More recently, a new approach known as hydrothermal carbonisation (HTC) has been experimented with for producing biochar from biomass under pressure at a relatively low temperature (Titirici et al. 2007; Lovett 2008). An optimised HTC process may be summarised as: catalysed HTC requires heating of the biomass dispersion under weakly acidic conditions (using citric acid as a catalyst) in a closed reactor for 5–14 hours to temperatures of less than 200°C. In addition, the HTC process requires wet biomass because effective dehydration only occurs in the presence of water, and the final C can be easily filtered from the reaction solution (Titirici et al. 2007). Although the pressurised HTC process requires little external energy for operation and appears simple and easy to operate, it has not been explosion-proofed and no commercial facility is commissioned yet.

As discussed previously, relatively abundant woody residues currently left in forests during harvesting operations can be potentially used for bioenergy or biochar production, or both, using pyrolysis or some other carbonation processes (Hall & Gifford 2008). One of the main reasons that most of these large quantities of woody residues have not been used is the very high cost of collection and transport to a facility for reuse. To minimise these costs, a mobile unit for making biochar should be developed for an on-site biochar production in recently harvested forest areas. Ideally, the biochar producing unit should be able to process woody materials with a large size and high moisture content.

Most of the biochar produced could be applied back to the forest from where the wood residues had been collected from. Such a biochar production and application system would require minimal external energy input because of minimal transport cost.

Biochar may be incorporated into the soil during site preparation (e.g., cultivation) before replanting. It may also be applied to the soil surface and used as mulch. Mixing biochar with soil can potentially improve soil nutrient levels and water retention capacity, as demonstrated in many studies conducted on agricultural soils (Glaser et al. 2002; Lehmann 2007). When it is incorporated into the soil, biochar is less likely to be lost through the erosion process. However, if very large quantities of biochar (e.g., hundreds of t ha⁻¹) are mixed with the soil, it may negatively affect the physical stability of trees in areas susceptible to wind damage. In addition, biochar mixed with soil can potentially stimulate native soil C decomposition as reported by Wardle et al. (2008).

Surface application of biochar causes minimal disturbance to the soil. It will also have little effect on soil structure and a potentially very high loading rate could be applied. Surface application can be made either before or after trees are planted, which makes application of biochar in plantation forests more flexible. On the other hand, surface-applied biochar is more likely to be lost in forests during fire events, though these are rare in New Zealand

Rumpel et al. (2006) investigated the soil organic matter loss caused by water erosion on steep slopes. They observed the preferential erosion of biochar compared to other types of soil organic matter. This was explained by the lighter weight of biochar and lack of mineral interactions following its formation during a fire. Therefore, biochar should not be applied to forests with steep slopes.

4.7.4 Priorities for future research

Although many studies have demonstrated that, in addition to C sequestration, biochar could potentially play an important role in helping soils retain water, nutrients, and may also support microbial activity that maintains and improves soil fertility. However, biochar research is still in its infancy. Much more research is necessary to get a better understanding of the full potential of biochar for C sequestration, particularly in forest soils. Further research in the following areas is required:

- Better quantification of the increased C sequestration in forest soils due to biochar application in order to qualify for carbon credits.
- Development of practical procedures for the collection of woody residues from forests.
- Development of a mobile and efficient pyrolysis unit that can achieve maximum biochar output with minimal external energy input.
- Optimisation of operational procedures for biochar production using woody residues.
- Evaluation of the effect of biochar and its application methods (surface versus incorporation) on mineralisation of native forest soil organic C.
- Evaluation of the impact of different biochar application methods on forest soil physical, chemical, and biological processes.
- Evaluation of the effect of biochar on development and growth of mycorrhizal fungi and associated soil C sequestration.
- Assessment of the impact of biochar on the biodiversity of forest soil ecosystems.
- Quantification of the response of tree growth rate and wood properties to biochar application.
- Evaluation of the effect a high application rate of biochar on water yield and quality from forests.

4.8 Current Best Estimates of Forest Soil Carbon Stocks and Change

4.8.1 Stocks and change under current land use/management (1990–2020)

There is no available information on the national areas of plantation forests under the different management groups identified above (or for any grouping). Nor are there comprehensive national estimates of carbon stocks under plantation forests on the various soil orders of New Zealand. In the absence of such data, we can only give the magnitude of stock changes that have been observed in response to various management practices under specific site and experimental conditions (from various field studies). These data are summarised in Table 8. Many of the studies are restricted to a single site, which limits the ability to apply the results at national or even regional scales. A further limitation is that in some cases

estimates of C stock changes are based on experimental treatments that are not representative of operational forest practice.

Table 8 Effect of forest management practices on soil carbon stocks

Activity	Variation	n ¹	Effect on soil carbon	Reference
Silvicultural practices	Stocking rate	1	Soil carbon reduced by ~ 10 t C ha ⁻¹ in 0–7.5 cm depth range with an increase in stock from 50 to 200 stems per ha, but little change above 200 stems per ha	Perrott et al. (1999)
Harvesting	Ground-based logging	1	Soil carbon reduced by 5 t C ha ⁻¹ in 0–10 cm depth range, but no change if soil sampled to 100 cm depth	Oliver et al. (2004)
	Hauler logging	1	No change in soil carbon	Oliver et al. (2004)
Site preparation	Spot-mounding	1	Soil carbon reduced by 4 t C ha ⁻¹ in 0–30 cm layer	Jones (2007)
	Line-ripping	1	No effect	Jones (2007)
Residue management	Residue removal	4	FH layer C reduced by 4.4 t C ha ⁻¹ after 8–16 years	Smaill et al. (2008a)
	Residue + forest floor removal	4	FH layer C reduced by 8.4 t C ha ⁻¹ after 8–16 years	Smaill et al. (2008a)
	Residue removal	1	Whole soil (L+FH+0-30 cm mineral soil) reduced by 6.9 t C ha ⁻¹ after 16 years	Jones et al. (2008)
	Residue + forest floor removal	1	Whole soil (L+FH+0-30 cm mineral soil) reduced by 10.2 t C ha ⁻¹ after 16 years	Jones et al. (2008)
	Residue burning	4	L and FH layer C reduced by 8.4 t C ha ⁻¹	Robertson (1998)
Site improvement	Repeated fertilisation	N 6	FH layer C increased by 5.8 t C ha ⁻¹ after 8–16 years	Smaill et al. (2008b)
	Ground-cover vegetation retention	1	Soil carbon in 0–10 cm layer increased by 10 t C ha ⁻¹ after 2 years	Jones, 2004

¹ The number of study sites used in the determination of soil carbon stock change.

4.8.2 Stocks and change for post-2012 mitigation options

The principle mitigation opportunities provided by forest management in relation to soil carbon are avoidance of practices that are likely to reduce soil carbon stocks, by:

- retention of harvest/thinning residues and forest floor litter on site;
- retention of a grass or weed cover during forest establishment, and post-harvest;
- avoidance of unnecessary soil disturbance (cultivation) during forest establishment;
- maintenance of soil fertility by fertiliser applications.

Of these opportunities, it is likely to be only the first two that are practically achievable as part of economically sound forest management. Retention of residues, and grass/weed cover, seem likely to reduce soil carbon losses by between 4 and 10 t C ha⁻¹. Creating a grass/weed cover post-harvest is likely to be particularly important when forests are harvested from hill country. Currently, it is the objective of Government to create incentives that result in new plantings of about 20 000 ha per year between 2012 and 2020. Using a value for mean soil carbon loss of 7 t C ha⁻¹ induced by failing to implement retention of residues and weed/grass cover implies small average annual losses between 2012 and 2020 of about 0.1 Mt CO₂ yr⁻¹—about 5% of New Zealand's present Kyoto commitment to emissions reduction.

The other two options listed above as potential mitigation practices are not likely to be practical to implement. That is, for the third option, the practice that shows some potential for soil carbon losses (spot mounding) is likely to occur irrespective of soil carbon considerations because of influence on tree biomass growth—and foregoing spot-mounding may in any event result in less total (forest plus soil) carbon gain at sites where spot-mounding is required to avoid growth limitations. In addition, for the fourth option, what stands would benefit from fertiliser application, and what carbon gains could be expected per unit of fertiliser applied, and at what cost would need to be assessed. The gains must also be balanced against possible nitrous oxide emissions from nitrogen-based fertiliser, and other possible environmental problems, such as nitrate leaching, need to be carefully considered.

The other soil carbon mitigation option considered here was the possible future biochar application; especially if it can be produced on-site from harvest/thinning residues together with the production of bio-energy. Although this has theoretical potential, it is already clear that a number of significant issues will require careful work to be completed before any net benefits can be reliably determined. Overall, at this stage, the net benefits of biochar application as a forest management practice remain unclear.

This is because of the following issues:

- Use of harvest residues for combined bio-energy/biochar production needs to be balanced against retention of residues to maintain soil carbon levels. The overall weight of evidence presently suggests that retention of the litter fraction on-site would be sufficient to retain soil carbon stocks, but this needs proper quantitative evaluation.
- Application of biochar to forest (and other) soils may promote more rapid decomposition of original in situ soil carbon stocks, leading at least to lesser gains than

otherwise anticipated. The interaction between biochar, original soil carbon stocks, and maintenance of soil carbon stocks and fractions by litter needs careful study. It should be anticipated such studies will be considerably complicated by the long-term nature of soil carbon stock-change, and the level of spatial heterogeneity that makes detection of small changes and slow trends (either positive or negative) problematic.

- Application of biochar may change soil properties in ways that may make it difficult to predict the net outcome: for example, it may lead to both improved water retention and enhanced water repellency; and to both positive and negative effects on nitrogen mineralisation rates. Careful studies will be required to determine the overall effects, and it may be difficult to reliably infer New Zealand-specific responses from overseas studies that have been conducted under different soil, climate and microbiological conditions.

4.9 Discussion

4.9.1 Implications of forecasts and scenarios

There are a range of forest management practices that can cause small but significant changes in mineral soil carbon. Whether such changes, if they occur, will need to be accounted in the future depends on the outcome of current international negotiations. If accounting of Article 3.4 activities become compulsory after 2012, or if any form of net-net accounting is introduced, then the effects of forest management on soil carbon stocks will need to be considered. Moreover, any changes in soil carbon stocks associated with forest management should already be reported under the UNFCCC.

Under future afforestation/reforestation programmes such as the ETS or PFSI aimed particularly at expanding total the post-2012 forest area, the most significant management-induced soil carbon change would likely occur if harvest residues are utilised for bio-energy and biochar production. The effect of spot-mounding may also need to be accounted, although whether this would be significant remains unclear due to the lack of data on the extent to which this is occurring. Mean losses of about 4 t C ha⁻¹ seem likely if spot-mounding is used, or up to about 10 t C ha⁻¹ if all harvest residues are removed from sites. If residue removal is restricted to larger woody material only, losses may be negligible—but this has yet to be determined in any quantitative manner.

For existing forests, significant amount of soil carbon may be lost during harvests, particularly in steeper hill country. Rapid establishment of a grass or weed cover is likely to mitigate such risks, and may also help compensate for losses of soil carbon stocks due to harvest residue removal. Maintaining a grass or weed cover during forest establishment also seems likely to reduce soil carbon losses associated with afforestation/reforestation. However, the extent to which this occurs has yet to be properly quantified, and whether it is an economic proposition remains an open question given that the immediate surroundings of young seedling need to be kept weed-free to avoid competition, and it can be very costly if weeding is to be applied so selectively.

4.9.2 Effects of information gaps/uncertainties on forecast/scenario reliability

It is not currently possible to quantitatively assess the effects of knowledge gaps and uncertainties on post-2012 soil carbon stocks and change under forest management practices.

It can only be concluded that there are two areas where there is a risk of soil carbon loss in the range of about 4–10 t C ha⁻¹: spot-mounding as part of planting practice, and removal of harvest residues from forest sites. At present, removal of harvest residues is not common industry practice, and thus risks of carbon losses will only arise if it becomes common for residues to be used for bio-energy and biochar production. Even with significant overall residue use, it is likely significant soil carbon losses would occur only exist if finer material were used as well. Biochar application as part of forest management may itself present some risks of lowering existing soil carbon stocks, and altering nitrogen availability, although overall it would seem likely to increase total soil carbon stocks.

Losses associated with spot-mounding appear to be at the lower end of the 4–10 t C ha⁻¹ range, and it is presently not known how widely this practice is applied. A further risk is that of significant soil carbon loss at the time of harvest, particularly in steeper hill country. Potentially, losses could be large, although much “loss” is likely to comprise transport to other areas (and to the sea) than genuine loss to the atmosphere, although for carbon accounting, currently, no distinction is made between loss and transport. Rapid establishment of a grass or weed cover is likely to mitigate such risks, and restrict the time window during which rainstorms may result in substantial erosion loss to about 6 months every rotation. Overall, however, given current industry practice, the extent of forests in hill country that are approaching harvest age, and the potential for large carbon losses, this could become a significant issue. Quantification, however, is not presently possible.

4.10 Conclusions and Recommendations

We have reviewed the New Zealand and international literature on the effects of plantation forest management on soil carbon stocks. This has revealed a number of important points, not only in relation to the nature of the effects themselves but also regarding our ability to use existing experimental results and data to make national estimates of soil carbon stocks and stock changes. The important points are:

- Forest management impacts are usually restricted to surface soil layers (i.e. the top 10 cm or so).
- Effects are highly spatially variable and site-specific although some general trends have been observed both among New Zealand studies and internationally.
- Key limitations of the existing body of work include:
 - a limited range of soil and site types investigated within New Zealand;
 - inconsistent methodologies (e.g., sampling depths, parameters measured, experimental design, nature of the treatments applied, sample preparation protocols);
 - frequent lack of bulk density measurements to accompany C concentration measurements;
 - an ad hoc nature of many studies, which has meant that most key management practice effects are yet to be fully investigated (e.g. no national inventory dataset);
 - uncertainty about the applicability of international results and data to New Zealand circumstances due mainly to site and management differences;
 - very little available New Zealand data on change in C stocks over time.

- Change in soil carbon stocks due to various forest management activities in New Zealand is usually reported (by a wide range of different studies) to be in the order of 4 to 10 t C ha⁻¹ (see Table 1).

General trends evident in New Zealand literature were that:

- tree stocking rate (only up to about 200 stems ha⁻¹) seems to have a positive relationship with forest floor C stocks, and a negative relationship with mineral soil carbon stocks;
- forest harvesting techniques that physically disturb the soil may cause the displacement of soil carbon but without causing significant carbon losses (other than on erosion-prone hill country);
- removal of harvest residues (particularly together with forest floor materials) tends to reduce C stocks of surface mineral soils, whereas the retention of residues at least maintains soil carbon stocks;
- some mechanical site preparation practices (usually involving physical disturbance and cultivation of the soil) can result in the loss of soil carbon;
- the application of fertiliser and the retention of a weedy vegetation cover can help maintain soil carbon stocks, and large fertiliser application rates tend to increase soil carbon stocks (with concomitant nitrous oxide emissions).

The review of available New Zealand datasets revealed there are some data that give, or could potentially give, an indication of the effects of many of the key forest management practices. However, these datasets tend to have several important limitations for this application. The limitations are that:

- almost all reported observations are experimental, meaning the results are obtained for specific circumstances that may not be representative of wider soil and site conditions or operational forest management;
- most studies were developed for purposes other than assessing effects on soil carbon so that the experimental design may not always be ideal for observation changes in soil-carbon stocks;
- carbon stocks were not usually measured to 30 cm (often to only 5 or 10 cm);
- bulk density often was not measured;
- they often cover a very limited geographical range;
- sampling and measurement protocols did not follow any consistent methodology.

The review has also shown that there is considerable potential for soil carbon stocks to be maintained or, in selected instances, to possibly be enhanced through the judicious selection and use of appropriate forest management practices. Nevertheless, there are corresponding

risks to forest growth and productivity in attempting to manage for soil carbon sequestration, and so these risks should be carefully assessed and managed relative to gains in biomass carbon.

At this stage, only very general values for expected soil carbon stock change following some key forest management practices can be given (see Table 8 above), in part because of some of the limitations listed above but also because no data are available on the areas of land in New Zealand currently subject to the various types of forest management. Information about the area under various management regimes is essential before forest management options could be included in national carbon accounts.

4.11 Present Status of Studies, Datasets, Analyses and Forecasts

Studies of the impacts of forest management on soil carbon are mostly based on long term field trials established by SCION to undertake research on forest management, sustainability, and nutrition. One trial has been established by Lincoln University. The value of these trials for soil carbon research is that they are able to provide information on the effect of management practices on soil carbon stocks. The trials are chiefly funded by the Foundation for Research Science and Technology, mainly under the SCION Forestry and Environment (PEEF) programme, but also by Future Forests Research Ltd under various projects. The PEEF programme is being re-bid currently, and the final outcome of this will have a strong bearing on the future of these trials. Neither the PEEF programme (in its current or future forms) nor the FFR projects specifically direct funds towards soil carbon studies. The Lincoln University trial is funded by a research trust.

Because the forest crop cycle lasts for about thirty years and effects of management practices on soil carbon manifest themselves over time, it is important that periodic re-measurements are made. As the trials have not been established specifically to determine the impact of management practices on soil carbon, periodic measurements are often lacking. Baseline soil data are commonly collected at trial establishment, but beyond the initial work, further sampling and analysis may or may not have been undertaken depending on the purpose of specific trials and whether or not associated studies have been undertaken. In the case of the LTSP I trial series, a further issue is the question whether samples had been collected before trials were lost through land use change. In a number of trials, baseline and subsequent sampling has been done, but samples have not been analysed for total C because of funding limitations. While some data on soil carbon from the trials are stored on shared drives, much is still held in spreadsheets by individual researchers. Therefore, it is recommended to undertake a stock-take of all long-term trials that have management treatments incorporated into their design, with a view to:

- collating all existing data and ensuring proper storage and backup of data;
- determining whether further sampling and analysis, or analysis of existing samples would contribute to an improved understanding of management impacts on soil carbon stocks and changes; and
- scheduling and undertaking further sampling and analysis as required.

New funding would be required to complete the above.

4.12 Key Uncertainties, Information Gaps and Research Priorities

Key research priorities include:

- The existing datasets are largely experimental, site-specific, and often do not include key C stock data to at least 30 cm. It is therefore warranted to develop a purpose-designed and comprehensive inventory dataset that covers a full range of soil and site conditions and key forest management practices. The priority is for better understanding of harvest residue management, and the contribution (and economics) of weed cover for the maintenance of soil carbon stocks. Further quantification of the impact of spot-mounding may also be appropriate once it has been determined how widespread this practice is.
- Development of consistent soil C stock measurement methods and protocols is required so that investigations under all land uses are directly comparable—with respect to inclusion of the coarse fraction, sample depth range, fine grinding, fine root processing, and sampling design (to take care of variability) issues.
- Determination of actual land areas under the various key forest management groups at a national level is needed to allow full carbon stock and stock change calculations to be made.
- Identification of appropriate soil C spatial prediction methodologies is required to allow extrapolation (and mapping) of soil C stocks from measurement sites to cover all forest areas (and potentially the whole country),
- More work is needed on understanding the mechanisms for soil C stock change caused by management impacts, land-use change, or climate change—including studies and modelling of C dynamics.
- The main opportunities for mitigation of soil C loss are in the areas of reduced soil disturbance at harvest, establishment and maintenance of a vegetation cover after harvest, and fertiliser application. More research is needed in all these areas over a range of sites and soils to allow results to be applied widely, and to quantify the actual magnitude of the mitigation opportunities.

4.13 Implications of Accounting and Mitigation Options for New Zealand's Post-2012 Net Position

If accounting of Article 3.4 activities becomes compulsory after 2012, or if any form of net-net accounting is introduced, then the effects of forest management on soil carbon stocks will need to be considered. However, the effects of forest management activities are likely to be small, being due mainly to spot-mounding activities. They are also likely to be very small in relative terms, given the much larger changes in forest biomass.

It seems highly unlikely that wholesale harvest residue removal, which appears to induce small but significant losses in soil carbon, would occur for bio-energy and biochar production if it resulted in an overall increase in net emissions.

However, the overall risks associated with a combination of removal of harvest residues, and application back onto forest soils of biochar, remain to be quantified—and long-term effects may prove challenging to determine.

5. Effects of Pastoral Agriculture on Soil Carbon

Anwar Ghani (AgResearch), Louis Schipper (Waikato University), Craig Ross (Landcare Research)

5.1 Introduction

Pastoral agriculture has been the dominant land use in New Zealand for nearly a century. According to Statistics New Zealand (2006), nearly 72% of the total land area of New Zealand is used for pastoral farming. Due to high economic returns from the dairy sector, the area of pastoral agriculture has been increasing in recent years, and more land has been converted from forest pasture for dairying. In addition, the intensity of dairy farming has also increased on existing farms, with greater stock numbers supported by higher fertiliser, water and feed inputs.

Understanding the impacts of farm management on soil carbon stocks is critical for predicting carbon stocks and potential changes for pastoral soils. If pastoral land use has positive effects on total soil C, then even small changes could have a significant influence on New Zealand's net carbon balance, as large areas are involved. However, if the effects of current management practice are negative for soil C storage, then New Zealand's GHG emissions liability could be increased substantially—depending on the accounting approaches agreed internationally for the post-2012 period.

There is considerable international debate about whether pasture and rangeland systems can be used as carbon sinks, or whether they are carbon sources—and depending on the type and intensity of management practices (e.g., Derner & Schuman 2007; Soussana et al. 2004). Soil testing has been used in pastoral systems for a long time to assess the fertility of the topsoil to help determine nutrient requirements for optimum pasture growth. However, measurement of soil carbon has not generally been part of this routine testing. In those instances where soil organic matter contents have been measured, it has almost always been limited to the topsoil only (0–75 mm). Databases on soil carbon contents in New Zealand pastoral soils are therefore somewhat limited, and very few data exist to the internationally accepted reporting depth of 30 cm.

The lack of interest in soil carbon in pastoral soils is the product of a long-held notion that carbon levels in temperate pastoral soils stabilise 15–20 years after conversion from other land uses, reaching pseudo-equilibrium. As a consequence, soil C research has been regarded as a low priority, and this is reflected in the limited number of pastoral soil C data sets that have been collected.

5.2 Datasets Reviewed to Determine the Effects of Agricultural Practice

The datasets reviewed in this part of the report comprise the 10 most significant New Zealand soil data sets that include measurements of soil carbon under pastoral agriculture. The initial purpose of most datasets was for characterisation of soils for nutrient-related research. With only a few exceptions, much less emphasis was given to record other aspects of the management, such as fertilisation, irrigation, liming, stocking-rate practices, that are now thought to be increasingly relevant to estimating the impacts of these management practices on soil carbon stocks. The datasets reviewed are:

- The National Soils Database (NSD; Landcare Research)
- NSD resampling dataset (Landcare Research, Waikato University)
- Bioindicator dataset (AgResearch)
- Ballantrae dataset (AgResearch)
- Whatawhata dataset (AgResearch)
- Tara Hills dataset (AgResearch)
- Winchmore dataset (AgResearch)
- 500+ soils dataset (Landcare Research)
- Arable pasture dataset (Crop and Food)
- Organics dataset (AgResearch)

To capture the information known about existing soil carbon datasets in a consistent format, including that known about the management regimes that are likely to have influenced soil carbon stocks, a questionnaire was designed and sent to agencies that hold the specific datasets. The questionnaire requested information on location, land use and management at the site, sampling depths, methods of carbon measurement, and biophysical attributes. It also requested information on any available management data that could be used to assist in the interpretation of data trends.

The Sections below summarise the data in the 10 datasets, together with (where possible) the implications of management for carbon stocks and stock changes. Details on each dataset, including that from the questionnaires, can be found in Appendix 2 of this Chapter.

5.3 Review of New Zealand STUDIES AND Datasets

5.3.1 Landcare Research National Soils Database (NSD)

The NSD is a 'point' database containing descriptions of about 1500 New Zealand soil profiles, together with their chemical, physical, and mineralogical characteristics. The information is obtained from excavated pits, usually up to 1.5 m deep but sometimes deeper, from which samples were obtained for chemical and physical analyses. Drilling to depths of up to 15 m was sometimes needed for sampling deep layers of volcanic ash.

The description of each soil profile pit comprises more than 200 soil attributes, either collected in the field or generated through subsequent laboratory analyses. Soil acidity, organic matter content, clay and silt content, toxicity, and phosphate retention are just a few of the many chemical attributes recorded. Soil physical parameters measured include soil drainage characteristics, depth to the water table, depth to an impermeable layer, gravel content, water-holding capacity, and clay mineralogy.

The database is held by Landcare Research. Much of the data is accessible via Landcare Research's soil portal, at:

http://soils.landcareresearch.co.nz/contents/SoilData_NSD_About.aspx?currentPage=SoilData_NSD&menuItem=SoilData.

5.3.2 NSD—deep profile re-sampling

At March 2008, this dataset consisted of 66 re-sampled soil profiles collected around New Zealand. Land uses sampled include dairy, and a range of “drystock” land uses; drystock including sheep, beef, deer, horses, dairy runoff, etc. An initial paper has been published documenting the results from the first 31 sites, where large losses of soil carbon are reported for some land-use classes (Schipper et al. 2007). Most sampled profiles were deeper than 60 cm, with many closer to 1 m in depth. The data are held in an Excel spreadsheet by Louis Schipper at the University of Waikato. There are plans to move these data into the National Soils Database.

Analysis of this dataset, and a comparison with data from archived NSD soil profiles, has demonstrated that:

- (i) dairying on flat-land with non-allophanic soils (19 profiles) has resulted in a significant loss of soil carbon (about $1.0 \text{ t C ha}^{-1} \text{ yr}^{-1}$) over the 20 years since first sampled;
- (ii) dairying on flat-land allophanic soils (13 profiles), “drystock” on flat-land non-allophanic soils (23 profiles), and “drystock” on flat-land allophanic soils (2 profiles) have not changed in soil C stocks; and
- (iii) “drystock” farming on hill country (8 profiles) has resulted in gains of soil C.

Overall, taking account of the estimated changes in soil carbon stocks, and the areas in hill-country and flat-land farming, it appears soil carbon stocks under pastoral land-use are static or slightly increasing. Such conclusions must, however, remain tentative at this stage given the limited sample size—although such a conclusion is consistent with the earlier analysis by Tate et al. (1997), based on the original NSD dataset.

5.3.3 AgResearch soil bioindicator dataset

In 1995, AgResearch started a FRST-funded project to better quantify soil biological and biochemical characteristics of pastoral soils. This project was continued for 5 years. Normal protocols of soil testing for the pasture sector were followed, i.e. soils were collected from 0 to 7.5 cm depth. Troughs, fence line, dung and recent urine patches were avoided during sampling. Soil samples were collected from pastures that were on ash (36 sites), sedimentary (40 sites) and pumice (26 sites) soils. Some soil samples were also collected from nearby cropping, forestry and market gardening sites to compare the soil biological and biochemical characteristics. Findings were reported in published papers (Ghani et al. 2003, 2007). In 2006 and 2008, some of same sites in the Waikato and Northland regions were re-sampled. Results of soil carbon levels from the Waikato soils under pasture land use showed:

- (i) that in comparison with 1995–96, the amounts of soil carbon in the re-sampled pasture sites had increased approximately 15% in the top 0–7.5 cm depth. Northland soils

showed some fluctuations in soil carbon levels but overall, there was no change in the mean values;

- (ii) Dairy pasture on allophanic soils tended to have less soil carbon than under sheep/beef grazed pastures.

5.3.4 Whatawhata archived soil samples

To determine whether soil carbon was being lost in hill country pastures, archived soil samples were analysed and compared with current data. The samples were collected between 1984 and 2006 from two slope classes (steep and easy) at the Whatawhata Hill Country Research Station. Soil samples had been collected from paddocks that were fertilised with six different loading rates of phosphorous (P; 0 to 100 kg ha⁻¹ yr⁻¹), the primary limiting nutrient for grass-clover pastures in these hill country farms. Soils are archived by AgResearch (Hamilton) and have been analysed by Louis Schipper (University of Waikato).

The range of P fertiliser loadings allowed us to determine whether P would regulate changes soil C and nitrogen (N). In contrast to expectations, there was no unidirectional change in C and N between 1984 and 2006, and the size of changes in C and N were not dependent on P loading rate. Other results were that (for sampling depths of generally 0–7.5 cm):

- (i) on average, soil C initially increased during the first 6 years of the trial at 0.27 % C yr⁻¹ (1.56 t C ha⁻¹ yr⁻¹) and 0.156 % C yr⁻¹ (1.06 t C ha⁻¹ yr⁻¹) on easy and steep slopes, respectively. Subsequently, soil C declined at –0.024 % yr⁻¹ for the easy slopes (not significantly different from 0) and –0.066 % yr⁻¹ (0.45 t C ha⁻¹ yr⁻¹) for the steep slopes. Similarly, % N increased between 1984 and 1990 at 0.025% N yr⁻¹ (144 kg N ha⁻¹ yr⁻¹) and 0.012 % N yr⁻¹ (82 kg N ha⁻¹ yr⁻¹) on easy and steep slopes, respectively.
- (ii) Post-1990, small but significant, losses of total N were measured on the steep slopes of 0.004% yr⁻¹ (27 kg N ha⁻¹ yr⁻¹) with no change on the easy slopes.
- (iii) Differences in pasture production are the most likely explanation for the changes in total C and N. After 1990, there was a decrease in pasture dry matter production in summer/early autumn (about 40% less) and declines in N-fixing clover abundance. Rainfall was greater before 1990 than after 1990 during these seasons, and it appeared that post-1990 pasture production was limited by moisture rather than phosphorus. This study has been submitted to a special issue of Biogeochemistry (Schipper et al. submitted July 2008).

5.3.5 AgResearch Winchmore long-term fertiliser experiment (WM1/1)

A phosphorous and sulphur (S) fertiliser experiment (Nguyen et al. 1989) was initiated at Winchmore in 1952 on grazed, border-strip irrigated pasture. The current treatments are:

- (i) Control, no fertiliser applied;
- (ii) 188 kg superphosphate ha⁻¹ yr⁻¹;
- (iii) 250 kg superphosphate ha⁻¹ yr⁻¹;

- (iv) 376 kg superphosphate ha⁻¹ yr⁻¹;
- (v) Sechura RPR plus elemental S, equivalent to P and S in a 250 kg ha⁻¹ yr⁻¹ superphosphate application.

Lime was applied to the site in 1949, 1950 (both 2.5 t ha⁻¹) and in 1972 (4.4 t ha⁻¹). The site has not been cultivated since 1950.

There was an initial increase in soil C in the P and S fertiliser experiment at Winchmore from 1952 to 1963 for the 7.5 cm depth sampled irrespective of superphosphate application (Nguyen & Goh 1990). This increase was associated with increased herbage production because of irrigation, pasture resowing, and liming, and followed the effective loss of C in topsoil when the paddocks were border-dyked (using a grader).

- (i) During the initial period from 1952 to 1963, soil C (0–7.5cm) increased from 2.7% to 3.7% in the fertilised treatments, which is equivalent to a sequestration rate of 0.9 t C ha⁻¹ yr⁻¹.
- (ii) For the next 30 years, soil C did not change significantly in treatments receiving superphosphate compared with the unfertilised treatment (Nguyen & Goh 1990).
- (iii) By 1993, researchers (Murata et al. 1995; Olsen 1994) found that there was no significant difference in soil C between any of the treatments. Average soil C stocks for the 188 kg superphosphate ha⁻¹ yr⁻¹ treatment were higher (but not significantly so) than for the 0 and 376 kg superphosphate ha⁻¹ yr⁻¹ treatments during 1993 to 2001 (mean soil C for 4 sampling dates: 3.95, 4.13, 3.94% C for 0, 188 and 376 kg superphosphate ha⁻¹ yr⁻¹ respectively; LSD_{0.05} 0.17%). This result could be consistent with under-utilisation of herbage by stock on the 188 kg superphosphate ha⁻¹ yr⁻¹ treatment—the stocking rate on this treatment was reviewed in 1996 and increased.
- (iv) Stewart and Metherell (1999a) found no significant differences between treatments in soil C for 0–10cm and 10–20 cm depths in 1997. There were also no irrigation or fertiliser effects on soil bulk density (Stewart & Metherell 1999a), so differences in soil C between treatments would be proportionately similar whether expressed as a percentage or on an area basis.
- (v) Comparisons have also been made with a nearby dryland, ungrazed “wilderness site” (Haynes & Williams 1992; Olsen 1994). There have also been studies of organic matter physical fractions, microbial biomass and ¹⁴Cages on this site.

5.3.6 AgResearch Winchmore long-term irrigation experiment (WM4/1)

The long-term irrigation frequency experiment (Rickard and McBride, 1986) at Winchmore was initiated in 1949 on border-strip irrigated pasture. Irrigation treatments were changed in 1953 and 1958, but dryland plots have never been irrigated, and the site has not been cultivated since 1958. Treatments maintained since 1958 include:

- (i) Dryland
- (ii) Irrigated at 10% soil moisture

- (iii) Irrigated at 20% soil moisture. Approximately 100 mm of water was applied per irrigation application
- (iv) Superphosphate has been applied at 250 kg ha⁻¹ to all treatments annually. Lime was applied in 1948 (5.0 t ha⁻¹), 1953 (1.9 t ha⁻¹) and 1965 (4.1 t ha⁻¹).

On the Winchmore irrigation experiment, analyses of soil C (0–7.5 cm) from around 1970 and from 1997 to 2001 have consistently shown a trend of highest C levels in the dryland treatment and significantly lower C levels in the most frequently irrigated treatment (20% soil moisture) (Metherell et al. 2002; Stewart & Metherell 1999a; Metherell 2003), despite an increase in herbage production with increasing irrigation frequency.

5.3.7 AgResearch Tara Hills long-term grazing experiment

A long-term grazing experiment on a steep, oversown tussock site at Tara Hills High Country Research Station in the semi-arid (precipitation of approx. 500 mm yr⁻¹) high country (910 m above sea level) of the South Island, New Zealand (Allan et al. 1992) began in 1978. Most production data was collected in the first 10 years, with the grazing treatments being maintained since then, but AgResearch sold the Research Station to Ngai Tahu in 2007. The site is steep (27°) and contains indigenous short tussock species as well as improved legumes and grasses from over-sowing. It has been fertilised with both P and S periodically since 1965. Treatments are continuous, alternating (two paddock system) or rotational (six paddock system) grazing with 1.9, 3.0 or 4.1 sheep ha⁻¹ during summer months. The experiment has a plot size of 1.7 ha and is un-replicated.

The initial soil sampling in 1979 was of upper-, mid- and lower-slope areas in the continuous treatment. In 1984 and 2003, intensive soil samplings of seven altitudinal strata within each plot were conducted. From 1996 to 1999, detailed studies of carbon cycling were conducted on the continuous and alternating grazing management treatments at three stocking rates. For statistical purposes the stocking rate by grazing management interaction is used as the error term, which gives a conservative assessment of statistical significance. For the 1984 and 2003 results, a stocking rate by grazing management interaction term was estimated from the interaction of two orthogonal transects, with the remaining interaction terms used as the error term (Allan 1985):

- (i) With pasture development, in a tussock grassland environment, soil C levels have in most treatments at least been maintained or possibly increased.
- (ii) In the 2003 soil sampling, higher soil C levels were found in the stock camp zones at the upper part of each paddock of all treatments, but an altitudinal trend had not been observed for soil C in 1984 (Allan 1985).
- (iii) In 2003, the lower stocking rate resulted in significantly higher soil C concentrations, primarily because of high soil C levels in laxly grazed areas in the lowest altitudinal strata of some low stocking rate treatments. Over all altitudes, the stocking rate effect was most pronounced in the continuous grazing management treatment, with the highest soil C levels found in the low stocking rate continuous grazing treatment, and the lowest soil C levels in the overgrazed high stocking rate continuous treatment.

- (iv) Although an effect of stocking rate was apparent in an initial pre-treatment sampling of the continuous treatment plots in 1979, and in the 1984 results (Allan 1985), the magnitude of the effect has increased with time. Similar trends were observed in two samplings in 1997, particularly in the surface 10 cm, although the effect did not reach statistical significance (Stewart & Metherell 2001).

5.3.8 AgResearch long-term (1975–2007) fertiliser and sheep grazing experimental site at Ballantrae

Two 10-ha farmlets, one having low (LF) and the other high (HF) fertiliser inputs, were established in 1975 on the Ballantrae Hill Country Research Station of the then DSIR and more latterly AgResearch. The LF farmlet received an average of 125 kg superphosphate (SSP)/ha/yr. The HF farmlet received an average of 625 kg SSP/ha/yr from 1975 to 1979, as well as 1250 kg lime in 1975 and 2500 kg lime in 1979. Since 1980, one HF farmlet has received 375 kg SSP/ha/yr, while the other has received nil. Since 1980, one LF farmlet has received 125 kg SSP/ha/yr, while the other has received none. The initial Olsen P levels of both farmlets was 5 µP/g soil and the retention was low (21–34%). The pH of the LF farmlet was 5.1 and 5.4 for the HF farmlet. Both farmlets were grazed with set stocked Romney breeding ewes. The stocking rate was initially 6 ewes/ha (1974) and this was increased in subsequent years in accordance with changes in pasture production.

- (i) Average annual pasture production from 1980 to 87 was 12.9 t DM ha⁻¹ yr⁻¹ for the HF farmlet and 8.4 t DM ha⁻¹ yr⁻¹ for the LL farmlet.
- (ii) Withholding fertiliser from the HF system resulted in a reasonably consistent decrease in pasture production of 4.6% p.a. from 1980 to 87. The decline in pasture production from withholding fertiliser from the low input system was much more erratic, but was, on average, 1.7% p.a. over the same period.
- (iii) Withholding fertiliser had little effect on the botanical composition of the pastures or the seasonality of pasture production. The performance of the farmlets receiving no fertiliser has continued to decline, with reversion to bushland a major issue on the LF
- (iv) Changes in soil C to a depth of 75 mm across these four farmlets and a number of other systems at Ballantrae was published by Lambert et al. (2000), indicating a statistically significant net loss of soil carbon of 200 kg/ha/yr over the last 10 years on both the LF and HF farmlets.
- (v) These four farmlets were sampled to two depths, 0–75 and 75–150 mm, with separate BD measurements in 2004, at a total of 72 sites. The sites covered 3 slopes and aspects. Only a preliminary analysis has been completed at this stage.

5.3.9 AgResearch long-term comparison of conventional and organic sheep and beef production

A long-term replicated farm systems study (1997–2007) examined changes in the biology of mixed-livestock systems associated with the shift to organic production. Two farmlets were managed using conventional farm practices (Con), and the two organic (Org) farmlets complied with the organic production standards of BIO-GRO New Zealand.

This study represents a world first: a long-term replicated farm systems study examining the changes in the biology of legume-based, mixed-livestock systems associated with the shift to organic production.

Soil and pasture sampling has been limited. The opportunity exists to complete a comprehensive comparison of the changes in soil C under conventional and organic practices under very controlled experimental conditions.

5.3.10 Landcare 500+ soil dataset

Between 1997 and 2001, Landcare Research conducted a programme to sample and analyse a large number of soils from the predominant New Zealand intensive agronomic land uses (dairy pasture, sheep and beef pasture, cropping and horticulture, plantation forestry and indigenous vegetation), encompassing all the major Soil Orders across New Zealand. This project was co-funded by MfE and Regional Councils. A strict sampling protocol was used where soils were collected from 0 to 10 cm depth (along a 50-m transect) at each site. It was planned that sites would be re-sampled on a regular basis (varying between 3 and 10 yrs depending upon land use) to monitor temporal trends in soil quality indicators. On-going sampling for individual Regional Councils has continued to the present on a cost-recovery basis, by Landcare Research. Soils are archived by individual Regional Councils, and Landcare Research. Data are held in Excel spreadsheets, and in reports to Regional Councils.

5.4 Datasets for Determining Pastoral Agricultural Soil Carbon: A Summary

The majority of the datasets on pastoral soils in New Zealand do not have information on soil carbon beyond 7.5 cm depth (Table 9). Most of the datasets that do have information on soil carbon beyond 7.5 cm lack accompanying information on pasture management practices which, as is increasingly realised, can significantly influence soil carbon levels (Lambert et al. 1998; Ghani et al. 2003; Metherell et al. 2008). There are no datasets with good information on both land management history and measurements of soil C stocks to 30-cm depth.

Most of the long-term pasture trials (Ballantare, Winchmore, Tara Hills, Whatawhata) maintained by AgResearch have good information on pasture management, but do not have information on soil carbon beyond the top 7.5–15 cm depth. Additionally, Tara Hills is no longer operated, and funding for continued soil monitoring at the others sites is intermittent, at best.

Landcare Research's National Soils Database (NSD) has already been essential for estimating baseline soil carbon stocks, from archived soil profiles that were initially sampled in late '70s to mid-'80s. Some of the NSD sites that were re-sampled between 2005 and 2008 (Schipper et al. 2007; Schipper, pers. comm.) will be useful for estimating a mean annual change in soil carbon stocks over the last 20 years. However, use of these datasets is still limited because of a lack of detailed information on the management history, i.e. on fertiliser inputs, effluent application, stocking rates, stock type, pasture species, and production levels. For some sites, this information may be able to be constructed—at least at a qualitative level. Also, because there were only two sampling periods (20 years apart), it not known whether changes in soil carbon stocks occurred over a short period, or more gradually and uniformly between sampling dates. Critically, it is not clear whether losses are still on-going.

While current assessment of C stocks for international C accounting purposes requires sampling to 30 cm, it is possible that eventually C changes much deeper in the soil profile will have to be considered. This is because changes, at least under intensive dairying on non-allophanic soils, seem to be occurring to at least 1 m depth (Schipper et al. 2007). At present, there are very limited data on New Zealand soil C stocks below 30 cm.

Table 9 Characteristics of soil carbon datasets relevant to carbon stocks in New Zealand pastures

Name of the dataset	Sampling depth*	Single vs. multiple sampling**	Bulk density measurements	Long-term monitoring	Basic land use information (e.g., dairy vs drystock)	Management (irrigation, fertilisation, stocking rates etc)	Site relocation (co-ordinates, or GPS location)	Soil archived samples
National Soil Database	a, b, c, d	A	Yes	Yes	Yes	No	Some	Yes
Resampled NSD	a, b, c, d	a, c	Yes	Yes	Yes	No	Yes	Yes
Bio-indicator	a	a, b, c	Yes	Yes (37 sites)	Yes	Yes	Yes	Yes
Ballantrae	a	a, b, d	Yes	Yes	Yes	Yes	Yes	Limited samples
Winchmore	a, b	a, b, d	Yes	Yes	Yes	Yes	Yes	Limited samples
Tara Hills	a	a, d	Yes	Yes	Yes	Yes	Yes	Limited samples
500+ soils	a, b	a, c	Yes	Yes (few sites)	?	Some	Yes	Yes
Whatawhata archive	a, b	a, b, d	Yes	Yes	Yes	Yes	Yes	Yes
Organic dataset	a	A	No	No	?	Yes	Yes	No

* a = 0.75 cm or 0–10 cm depth, b = 7.5–15 or 10–20 cm depth, c = 15–30 or 20–30 cm depths, d = includes some deeper samples

** a = single sampling, b = multiple sampling with in a year, c = two sampling points separated by several years, d = yearly sampling for several years

Long-term = more than 10 years, Short-term = less than 10 years

5.5 Tussock Grasslands

5.5.1 Introduction

Tussock grasslands cover about 3.3 Mha of New Zealand, with the largest areas being in the South Island High Country and Otago uplands (Fig. 3).

This review concentrates on the tussock grasslands that are farmed for livestock grazing or have been extensively grazed by sheep in the past but are now retired from grazing to the Conservation Estate. The soil carbon stocks in tussock grasslands in the non-farmed alpine and sub-alpine zones in both North and South Islands are assumed to be more or less at steady-state, except for the effects of soil erosion (being covered in a later section) and grazing by feral animals (mainly deer and rabbits, but also hares, chamois, and thar). No information was found during this review on the effects of feral animal grazing on soil carbon stocks that might relate to the Kyoto Protocol's first commitment period (2008–2012).

5.5.2 Carbon stocks in tussock grasslands

Tate et al. (1997) and Carswell et al. (2008) report the carbon stocks in the tussock grasslands given in Table 10. Tate et al. (1997) also presented soil carbon data for high-country and upland yellow-brown earths (Brown Soils) that were mainly under tussock grasslands (Table 11). Those data were based on a number of studies by Lincoln University, DSIR Soil Bureau, DSIR studies, and the National Soils database. However, it is known the list in Table 11 is an incomplete inventory of the available data from other College/University studies. For example, Ross (1971) reported average values of soil carbon for soil profiles to 57 cm depth of 148.7 and 162.3 t C/ha for shady and sunny aspects respectively, for a Kaikoura steepland soil under snow tussock. Cuff (1973) found average values for soil profiles to 40 cm depth (one to 25 cm depth) of 96.3 and 72.7 t C/ha for shady and sunny aspects of Hurunui steepland soils at three South Canterbury sites under tussock grassland. There were insufficient resources in this review to complete a comprehensive search for data on soil carbon under tussock grasslands from other Lincoln College/University or other university (Otago, Canterbury, Victoria) theses not reported here.

Other soil carbon data for tussock grasslands is recorded in Hewitt and McIntosh (1996), Nordmeyer (1997) and O'Connor et al. (1999), as well as reports and papers related to land use changes discussed in the following sections. The Hewitt and McIntosh (1996) report on *Soil Organic Matter in the South Island High Country* gives information on % soil carbon levels but not quantities in t/ha, except for the effects of grazing, oversowing and fertilizer as discussed later. Nordmeyer (1997) reports biomass and soil C for a number of snow- and short-tussock grassland sites (Table 12). He concluded that soil C reaches maximum values of about 200 t C/ha under snowgrass in cool, moist alpine grasslands. Induced short-tussock/browntop/hieracium systems contain 50–100 t C/ha soil carbon in stoney soils and up to 130 t C/ha in soils with deep loess accumulation. O'Connor et al. (1999) recorded biomass and soil carbon quantities 0–20 cm for nine localities with tall tussock grasslands (Table 13) and under various stages of degradation at three locations, the latter study including 20–40 cm soil C data (Table 14). Biomass ranged from 46 to 87 t /ha and 0–20 cm soil C from 100 to 163 t C/ha for the nine locations and 103 to 162 t C/ha for total biomass and soil (0–40 cm) for the degraded grasslands study.

Figure 3 Tussock grassland areas mapped from the Vegetation Cover Map based on Newsome (1987) data. The areas should be regarded as approximate only since vegetation cover will have changed somewhat since the late 1980s.

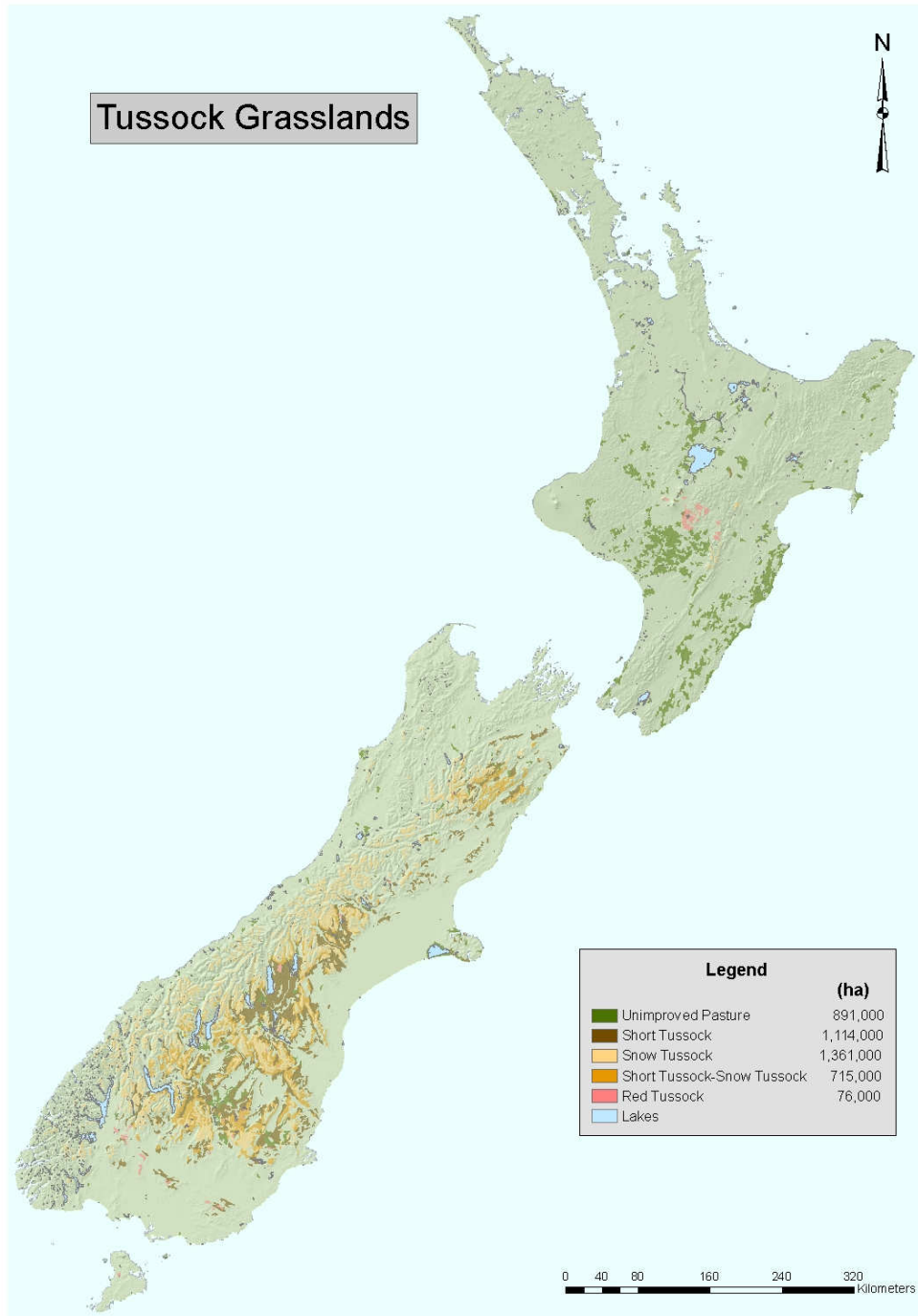


Table 10 Carbon stocks for vegetation classes in the Vegetation Cover Map for New Zealand, from Tate et al. (1997) and Carswell et al. (2008)

Vegetation cover	Code	Area (10 ³ ha)	Biomass (t C ha ⁻¹)	Total Carbon (Mt C)	Soil C (t C ha ⁻¹)	Total Soil C (Mt C)
<i>Grassland</i>						
Short-tussock	G3	1116	10.8	12	144.6	161
Snow-tussock	G4	1361	27.2	37	134.4	183
Short and snow tussock	G4	712	19.7	14	159.3	113
Red tussock	G6	80	22.5	1.8	171.5	14
<i>Grassland - Scrub</i>						
Tussock and sub-alpine scrub	GS4	958	21.9	21	137.8	132
Grassland and dracophyllum scrub	GS5	55	23.6	1.3	143.8	8
Grassland and matagouri	GS7	520	8.1	4.2	128.4	67
Grassland and sweet brier/matagouri	GS8	230	7.8	1.8	139.9	32
<i>Grassland – Forest</i>						
Tussock and beech forest	GF5	100	110	11	144.5	14
Tussock and podocarp-broadleaf-beech forest	GF6	10	160	1.6	131.4	1

Table 11 A comparison of soil carbon estimated for high-country and upland yellow-brown earths from a number of sources, including the National Soils database (from Tate et al. 1997). * The numbers of profiles recorded for the database means are those for which acceptable 0–1 m carbon estimates could be obtained. Numbers for 0–0.25 m carbon estimates are given in parenthesis

Soil Set/Series	No. of profiles*	Mean carbon (t C ha ⁻¹)		Source
		0-0.25 m	0-1m	
Craigieburn	4	82	132	Bruce & Childs (1981)
Kaikoura	6	72	186	Archer & Cutler (1983)
Katrine	2	69	129	Webb et al. (1986)
Mary	4	67	133	Webb (1976)
McDonald	6	111	266	Webb (1976)
Puketeraki	5	77	196	NZ Soil Bureau (1968) Ives 1970); Harvey (1974)
Te Koa	5	59	153	Ives (1970); Harrison (1982)
Tekapo	10	95	218	Webb (1976); Webb, pers. comm.
Upland ybe (mean)	3 (9)	97	243	National Soils Database*
High-country ybe (mean)	3 (5)	115	144	National Soils Database
Combined mean	6 (14)	103	184	

*The National Soils Database has soil C information for 118 sites (sampled soils) under tussock grasslands.

Table 12 Carbon in alpine grasslands (Kaikoura soils), hard tussock/hieracium systems (Tekoa soils) and dry inland basins (Forks soil) (from Nordmeyer 1997)

Reps	Vegetation	Biomass C	Total Soil C 0-60 cm		Ecosystem C
			----- (t C ha ⁻¹) -----		
1	Snowgrass	No data	79.0	-	-
1	Snowgrass	No data	103.5	-	-
1	Snowgrass	11.3	191.3	202.6	202.6
3	Snowgrass	13.9	189.6	203.5	203.5
1	Snowgrass	No data	234.2	-	-
3	Snowgrass	12.1	156.8	169.2	169.2
3	Tussock/hieracium	9.5	98.2	107.7	107.7
4	Hieracium/tussock	7.1	69.5	76.6	76.6
4	Riser hieracium	3.2	45.6	48.8	48.8
2	Hollow hieracium	5.1	89.6	94.7	94.7
4	Hieracium/tussock	8.7	119.9	128.6	128.6

Table 13 Biomass (t C ha⁻¹) of tall tussock (*Chionochloa*) grasslands and soil carbon (t C ha⁻¹) in the 0–200 mm layer at nine localities in South Island high country (from O'Connor et al. 1999)

Location	Biomass (t C ha ⁻¹)	Soil C 0-200 mm (t C ha ⁻¹)
Craigieburn Mountains		
<i>Chionochloa pallens</i>	74.0	86.7
<i>C. macra</i>	50.8	80.9
<i>C. rubra</i>	68.0	88.1
Hakatere Basin		
<i>C. macra</i>	38.7	61.7
<i>C. rigida</i>	72.3	90.6
Tekapo, Mackenzie Basin		
<i>C. rubra</i>	46.2	74.8
Old Man Range, Otago		
<i>C. macra</i>	66.2	95.2
<i>C. macra/rigida</i>	48.7	52.5
<i>C. rigida</i>	87.2	63.2

Table 14 Weights of C in biomass and soil pools at different stages of ecological degradation for three Canterbury high country localities (from O'Connor et al. 1999)

Locality, and Total C	Climate	Ecological Stage	Biomass C			Soil C 0-40 cm		
			----- (t C ha ⁻¹) -----					
Craigieburn	Humid subalpine	Tall tussock grassland	25.3	115.4	140.7			
		Weedy short grassland	10.7	92.0	102.7			
		Degradation loss			38.0			
Hakaterere	Moist sub-humid	Tall tussock grassland	36.2	146.6	182.8			
		Mixed tussock grassland	17.6	144.6	162.2			
		Weedy tussock grassland	5.9	154.8	160.7			
		Degradation loss			22.1			
Tekapo	Dry sub-humid	Short tussock grassland	10.9	123.6	134.5			
		Weedy short grassland	8.7	106.4	115.1			
		Degradation loss			19.4			

In summary, the range of biomass carbon in the tussock grasslands is about 6–36 t C ha⁻¹ and for soils to 1 metre or profile depth, 49–159 t C ha⁻¹.

5.5.3 Land use effects on carbon stocks in tussock grasslands

General Studies

Tate et al. (2005), reported changes in soil organic C resulting from land use and management changes from 1990 to 2000 compared with soil C stocks under improved grasslands as a baseline. For unimproved grassland (including tussock), Tate et al. (2005) reported changes of -0.3 ± 2 , 3 ± 2 , and 2 ± 6 t C ha⁻¹ for 0–0.1, 0.1–0.3, and 0.3–1 m depths, respectively. Although the standard errors are high compared with the means, the data suggests that unimproved grasslands have higher total soil C levels to 1 m than improved grasslands.

The data in O'Connor et al. (1999; Table 14) show losses through degradation of tussock grasslands of 38, 22, and 19 t C ha⁻¹ for combined above-ground biomass C plus soil C to 40 cm depth. Carswell et al. (2008), using the data from McIntosh (1997), estimated (assuming biomass is 50% C) the amounts of carbon in a degradation sequence from mountain beech forest through tussock grasslands to herbfield dominated by *Hieracium*. Excluding the beech forest, the C in biomass was: tall-tussock grassland biomass was 32–35 t C ha⁻¹, in short-tussock grassland 11 t C ha⁻¹, and in herbfield 1–2 t C ha⁻¹.

A Waikato University PhD project by Emily Weeks, based at Landcare Research, Palmerston North, on 'Intensive development of New Zealand's tussock grasslands: rates of change, assessment of vulnerability, and priorities for protection' will provide further relevant information on carbon stocks in tussock grasslands.

Erosion

Erosion in the tussock grasslands by wind and water, enhanced by frost heave, is a significant factor affecting carbon stocks. This is being covered in the review of soil erosion that appears elsewhere in this report.

Grazing / retirement and reversion to woody vegetation and forest

Studies on plots excluded from grazing for 15 years by Gregg (1964) recorded a combined biomass C and soil C (to unspecified depths that included 9 cm of the subsoil) increase of 35 t C ha⁻¹ (Table 15).

Table 15 Effects of grazing and intermittent burning, and exclusion from grazing, on the carbon stocks of snow tussock grasslands (Gregg, 1964)

	Biomass C	Soil C	Total C
	----- t ha ⁻¹ -----		
Snowgrass that represented pre-European landscape	59	136	195
Continuous grazing and intermittent burning for approximately 85 years followed by 15 years spelling	28	122	150
Continuous grazing and intermittent burning for approximately 100 years	15	100	115

Basher and Lynn (1996), in a follow-up study of two snow tussock sites that included the Gregg (1964) study exclusion plot, found no significant differences in soil C (0–15 cm) between plots excluded from grazing compared with grazed sites. The Starvation Gully site, studied by Gregg (1964), had 87.9 ± 6.1 t C ha⁻¹ inside the enclosure plot, compared with 92.0 ± 0.25 t C ha⁻¹ in the adjacent grazed land. On another site at Cloudy Knoll they recorded 77.5 ± 4.0 t C ha⁻¹ inside the enclosure plot and 67.0 ± 3.8 t C ha⁻¹ outside. The limitation of these results is the shallow sampling depth of 15 cm.

The review by Carswell et al. (2008) records that some areas of tussock grasslands are tending to woody vegetation, particularly with the prevention of fire (see next section) and where browsing or grazing animals are present. Plant succession to a continuous cover of taller woody species may take hundreds of years, especially in low rainfall areas. Reversion of tussock grasslands to forest, with retirement from grazing and periodic burning, is likely to take many decades. The effects of slow reversion from tussock grasslands to woody or forest vegetation on biomass and soil C, particularly on land now in the Conservation Estate, will be insignificant over the Kyoto first commitment period. In the longer term, these vegetation changes might need to be considered.

Burning

A review of the effects of burning on tussock grasslands by Basher et al. (1990) estimated potential fire-induced losses of soil carbon (0–8 cm) averaging 37 t C ha⁻¹ (range 16–130). Allen et al. (1996) report on the effects of fire on vegetation but did not present any carbon or biomass carbon data. Ross et al. (1997) found losses of 1.2% and 1.1% from the 0–2 cm layer of a Carrick soil, on two sites 1.5 and 2.5 years, respectively, after burning snow-tussock (*Chionochloa rigida* ssp. *Rigida*) grassland on the Old Man Range in Central Otago.

The effects of burning on carbon stocks have been integrated with the studies reported in the preceding sections on general degradation and grazing of the tussock grasslands. The general conclusion is that, as well as reductions in biomass carbon from burning, soil carbon also declines.

Oversowing and topdressing

Studies reported in Hewitt and McIntosh (1996) show an increase in 0–7.5 cm tussock grassland soil C of 1.5% from oversowing and topdressing. A sequence of sites from grazing only, no grazing for nine years, no grazing with oversowing, and no grazing with oversowing and superphosphate fertilizer, showed increasing amounts of biomass (about 4, 11, 16 & 26 t C ha⁻¹, respectively) and soil C (0–7.5 cm) (43, 50, 50 & 60 t C ha⁻¹, respectively). These data show that oversowing and topdressing tussock grasslands are likely to considerably increase both above- and below-ground carbon stocks.

Hieracium

The data presented in the previous sections for degraded tussock grasslands and by McIntosh and Allen (1993) and McIntosh et al. (1995) indicate significant reductions in biomass C for *Hieracium*-dominated areas compared with areas with a dominant tussock cover. However, soil studies show a build-up of soil C (by 0.7–1.2% C; 0–7.5 cm layer) under *Hieracium*, compared with adjacent bare ground—although this build-up is unlikely to equate to the levels of soil carbon found under equivalent tussock grasslands.

5.6 Mitigation Opportunities for Pastoral Agricultural Soils

5.6.1 Assessment of mitigation options

Some potential mitigation approaches to minimising soil carbon losses under pastoral agricultural practices are given in Table 16. However, it remains difficult to be certain that such approaches will succeed because it is currently unclear why some pastures are losing soil carbon, and others gaining it. Once there is a mechanistic understanding of soil carbon changes in pasture systems, mitigation practices can be better proposed and tested.

Present indications are that some mitigation should be possible because “dry-stock” farming on flat land does not seem to result in soil carbon loss, and hill country pastoral farming areas in the North Island appear to be gaining considerable soil carbon. However, these apparent trends in existing datasets should be treated with considerable caution, because they are based on samples from relatively few sites. Whether the losses that are apparently occurring under intensive dairying can be mitigated remains an open question, given the present lack of mechanistic understanding. It is also not entirely apparent that the losses recorded in the NSD deep profile re-sampling dataset are occurring nationwide (see the results from the LMI dataset in Section 5.2, which apparently run counter to the results from the NSD re-sampling work).

It may be tempting to suggest that the difference observed to date in carbon stocks between the different farming systems is due to direct physical impacts of stocking intensity. However, losses could also be due to differences in pasture management. For example, there is anecdotal evidence that intensive use of strip grazing under some dairying regimes may substantially reduce net photosynthesis and thus total pasture production, possibly also increasing soil respiration, resulting in a net loss of inputs to the soil, and thus lead to a loss of soil carbon. It also remains uncertain whether the results observed to date are typical of a wider range in sites and/or soil types. Much work remains to be done before definite conclusions can be drawn on the effect of land management and intensification on soil carbon stocks.

Table 16 Possible approaches to mitigating likely soil carbon loss in intensively-managed pastoral agricultural systems

Proposed mitigation technique	Rational for increased soil C	Practicality/drawbacks
No-till, minimum till pasture renewal	Avoids potential one-off losses of C associated with cultivation. However, the magnitude of losses, if any, associated with pasture renewal is presently unknown	Currently practised, possible expansion; requires specialist equipment, and may lead to weed-control problems
Optimising stocking rate	Reduce off-takes of C, maximise inputs to the soil	Can be done, but likely to lower production and economic costs could be high
Adding biochar	Slowly degrading C source added	Difficulty incorporating it into soil; distance between sites of biochar production and site of incorporation
Alternative grazing management	Maximise photosynthesis, minimise respiration	Would require re-design of grazing management systems, may have co-benefits of increased production

5.6.2 Environmental co-benefits and risks

While it is not clear what management practices within pastoral agriculture are leading to changes in soil carbon, identification of management practices that promote carbon storage would have considerable co-benefits. Soil carbon is at the centre of maintenance of soil quality, providing a wide variety of ecosystem services including nutrient and water storage, a food base for soil biology, regulation of other biogeochemical cycles (including nitrogen) and maintenance of soil structure.

These co-benefits contribute to maintaining production but also to protection of off-site environments, e.g., minimise nitrate leaching. If soil carbon losses continue in flat land dairy systems some of these functions could be compromised, requiring increased external inputs to maintain production and environmental benefits.

5.7 Current Best Estimates of Pastoral Agricultural Soil Carbon Stocks and Change

5.7.1 Carbon stocks and change

For accounting under the Kyoto Protocol, and reporting under the UNFCCC, it is good practice to determine soil carbon stocks to a depth of 30 cm. This depth of soil sampling is particularly relevant to the cropping soils that form the bulk of the global area under agricultural land use because this often represents the cultivation depth. At present, New Zealand is not required to account for changes in soil carbon stocks under agricultural land use. However, this is likely to become a requirement if New Zealand were to adopt

accounting under Article 3.4, or if any form of net-net accounting becomes mandatory, post-2012. Moreover, either Article 3.4 or future net-net accounting requires that the carbon stocks existing under agricultural land-use be determined for 1990 to provide a baseline for comparison of accountable change.

Tate et al. (2005) estimated total soil C stocks in grazing land for 1990 to be 1480 ± 58 Mt C for the top 30 cm. Datasets reported here will not, in general, be able to contribute to refining this figure because it appears likely, though not yet confirmed due to sample-size limitations, that intensification of land management since 1990 has changed soil C stocks for some land-use/soil/climate combinations.

Recent findings (Schipper et al. 2007; Parfitt & Schipper, pers. comm. 2008) drawn from the re-sampling of soil profiles at NSD sites (see above; Landcare National Soils Database—deep profile re-sampling), suggest some large changes in total soil C for some land-use/land-form combinations (Table 17). There appear to be large, statistically significant losses of C in dairying on flat land (-1.2 ± 0.3 Mt C yr⁻¹), although there may also be gains in North Island hill country ($+3.4 \pm 1.3$ Mt C yr⁻¹) over an approximately 20-year period. The figures for soil C change are for sampling up to a metre depth, rather than just the top 30 cm, because this is the depth over which change seems to be occurring under New Zealand conditions.

Table 17 Preliminary scenario for total carbon changes for New Zealand pastures under different broad soil type, land use and land form combinations, scaled on the basis of re-sampling to date of NSD soil profiles. This table is largely given for illustrative purposes, as the number of soil profiles sampled remains very small and uncertainty consequently large. At present, there is no statistical justification for concluding the total annual change in mean soil carbon stocks on land under pastoral agriculture is significantly different from zero, requiring no change to the earlier conclusions of Tate et al. (2003).

Land Category	Land Form	Land Use	No. of profiles	Area of pasture (Mha)	Mean Change (t C ha ⁻¹ yr ⁻¹)	SE (t C ha ⁻¹ yr ⁻¹)	Total Change (Mt C yr ⁻¹)	SE (Mt C yr ⁻¹)
NZ	Flat	Non-dairy and dry cows	18	3.9	-0.6	0.3	-2.5	1.3
Volcanic	Flat	Dairy	13	0.7	-0.2	0.5	-0.2	0.4
Non-volcanic	Flat	Dairy	14	0.8	-1.5	0.3	-1.2	0.3
North Island	Hill	Non-dairy and dry cows	12	3.1	1.1	0.4	3.4	1.3
South Island	Hill and tussock	Non-dairy and dry cows	12	2.6	-0.1	0.2	-0.2	0.6
Totals			69	11.1			-0.6	2.0

Uncertainty remains high because only a relatively small number of sites have been sampled to date. Tate et al. (2003) had concluded that soil carbon stocks under pastoral agriculture were not changing. At present, given the limited amount of information available and its associated variability, and given that some pastoral systems appear to be gaining carbon while others may be losing carbon, there is little justification for concluding that the total annual national change in mean soil carbon stocks on land under pastoral agriculture is significantly different from zero.

If changes in soil carbon with land management, under the same land use, have to be accounted in the future, a very large effort may be needed to improve estimates of soil carbon stocks and change if New Zealand wished to move beyond considering national stocks to be, overall, unchanging. Accounting quantitatively and defensibly for changes in soil carbon stocks would be challenging, as changes per hectare during a given accounting period are not likely to be large and, conversely, the areas involved are large and heterogeneous. Accounting would at the very least require a substantial sampling programme to:

- establish current soil carbon stocks more precisely;
- locate and sample areas representing 1990 management conditions to strengthen estimates of baseline stocks;
- model and validate rates of average change, drivers of change, and the time-dependence of change.

5.7.2 Stocks and change under current land use/management (1990–2020)

Based on current published data, Metherell et al. (2008) summarised the broad effects of land management on soil carbon stocks under New Zealand pasture, as shown in Table 18. Most of the observations reported in the table are based on soil carbon measured over only 7.5 cm depth. As Table 18 shows, a number of management practices apparently result in more than one outcome for soil carbon stocks, presumably due to the interaction of the particular practice—or the range in management intensity—with soil type and climate. Overall, it is apparent that:

- increased stocking rates appear to have negative effects on soil carbon, possibly due to loss of total biomass production, with reduced inputs to the soil carbon pool, and enhanced respiration, although exact mechanisms have yet to be established;
- impacts of fertiliser inputs, irrigation and pasture developments on soil carbon can be both positive and negative, depending on the baseline soil carbon stocks, other soil physical and chemical properties, and the prevailing climate.

Many of New Zealand's agricultural management practices are unique, particularly when seen in the context of our soils and climate. It is therefore difficult to relate overseas studies to the New Zealand situation, as by far the majority of international work is related to behaviour of cropping soils, rather than to soils under pastoral agriculture. From international work, it can only be concluded that the following broad trends could be expected.

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- It is well established that when soils are cultivated some loss of soil carbon occurs due to accelerated respiration of soil carbon by soil decomposer organisms.
- Decreased returns of above-ground biomass through harvesting and removal from the site generally lead to soil carbon losses.
- When arable land is under pasture rotation, carbon accumulates to some extent in the soils.

Taken overall, there is currently little evidence to suggest that any net change in soil carbon stocks is occurring in New Zealand under pastoral agriculture—consistent with the earlier national-scale analysis by Tate et al. (2003). Although recent studies suggest it is possible that stocks are increasing in hill country, and decreasing under intensive dairying on non-allophanic soils, sample sizes in these studies remain small. Conclusions must therefore remain tentative at this stage.

Table 18 A qualitative evaluation of the effects of land management on soil carbon in New Zealand pastoral soils (after Metherell et al. 2008)

Management	Effects on soil carbon
Pasture development	+, -
Arable to pasture	+
Cultivation	-
Fertiliser inputs	+, =, -
Irrigation	=, -
Intensification (stocking rates, pasture utilisation)	-
Sheep to dairy	-, ?

Increase in soil carbon (+); decline in soil carbon (-); no change in soil carbon (=)

5.7.4 Stocks and change for post-2012 mitigation options

At the present time, there is not sufficient evidence to provide a defensible basis for recommending particular mitigation practices for reducing losses, or for enhancing, soil carbon stocks under pastoral agriculture. At best, the following risk factors that may result in lowered soil carbon can be identified—although it remains an open question when or where such risks might be realised under New Zealand conditions:

- Intensive dairying may result in small soil carbon losses on non-allophanic soils, which would be consistent with lowered below-ground inputs due to reduced production levels.
- If soils are cultivated as part of forage cropping to support pastoral agricultural systems, it can be expected that small losses of soil carbon may occur. However, these are likely to be recovered provided the cropping is temporary and the area returned to grassland (see also Section 5).

- Cut-and-carry systems, which have recently begun to be considered for high-end production systems, could be expected to reduce inputs to soils under pastoral systems and so result in at least an initial reduction in soil carbon stocks.

5.8 Discussion

5.8.1 Implications of forecasts and scenarios

For New Zealand's pastoral agricultural soils, current data remain consistent with the view provided in national carbon reporting to date: that at the national scale, soil carbon stocks are static. This situation appears likely to hold from 1990 to 2020. However, it appears there may be some risks that increased intensive dairying may reduce soil carbon stocks somewhat, although current management may also be increasing soil carbon stocks in hill country—but neither of these trends has yet been established with sufficient certainty to be in any way definitive, and both will require a very substantial additional effort to prove.

5.8.2 Effects of information gaps/uncertainties on forecast/scenario reliability

At present, there remains a very significant lack of data on the magnitude and rate of change in soil carbon stocks with pastoral agricultural land management practice. It seems likely that total national soil carbon stocks are invariant, but significant local change could be occurring, both negative and positive, depending on the interaction between management intensity, soil type and climate. The difficult issue is that although changes may be small, and thus very difficult to detect due to soil heterogeneity, the areas involved are very large. There is thus a definite risk that significant losses could be occurring, or, conversely, that there could be substantial gains that are presently neither reported nor accounted. Should accounting of soil carbon change become mandatory post-2012, a much greater effort will be required to quantify soil carbon stocks and change.

5.9 Conclusions and Recommendations

5.9.1 Present status of studies, datasets, analyses and forecasts

New Zealand has relatively good data on soil carbon stocks in the topsoil (c. 0–7.5 cm) of pastoral agricultural. However, there are very few datasets with data below these depths: only those datasets in the NSD, and NSD deep profile re-sampling, datasets are sufficient to allow analysis to the 30 cm depth required under international carbon accounting conventions. Based on the NSD data, Tate et al. (2003) concluded that national average soil carbon stocks under pastoral agriculture were at steady-state, and thus unchanging over time. Recent work by Schipper et al. (2007) has presented results that, while agreeing with the conclusion of Tate et al. (2003) overall, nonetheless indicate that small losses, and gains, respectively, are apparently occurring under different management regimes.

- (i) For sites sampled to date, small, statistically significant losses of soil carbon have occurred over the past 20 years for dairying on non-allophanic soils. If these losses are widespread, and continuing, they are large enough to be of concern from the perspectives of maintaining soil quality, the sustainability of dairy farming, and the effect on New Zealand's carbon balance. Such losses would be accountable as part of the national net carbon balance if future accounting will be required under Article 3.4 or any form of net-net accounting.

- (ii) Studies to date have also indicated there have been small, statistically significant gains in soil carbon in hill country over the past 20 years, that, if they could be better quantified, would be accountable as part of the national net carbon balance if future accounting is required under Article 3.4 or any form of net-net accounting.
- (iii) The mechanisms for the above changes in soil carbon are not yet clear, and a number of hypotheses are being examined utilising the expertise of several research organisations. If the mechanism controlling carbon gains in hill-country soils can be better understood, there may be potential for proposing approaches to improved soil carbon sequestration. However, it needs also to be noted that the dataset sample sizes are neither large nor geographically representative, and so may yet contain biases.

5.9.2 Key uncertainties, information gaps and research priorities

The major limitation of New Zealand soil carbon datasets for pastoral soils is that the data were never collected for the purpose of characterising soil carbon for accounting purposes. Until recently, it was accepted that pastoral soils accumulated soil organic carbon during development before reaching pseudo-equilibrium. Obtaining other than topsoil data (c. 7.5 cm) was therefore not considered to be of importance until the Kyoto Protocol was adopted. Consequently, most existing data are inadequate for accounting purposes. Moreover, even for those few datasets where depths below 7.5 cm were sampled, information on land management was not usually recorded. As it is increasingly apparent that land management practise have at least the potential to significantly affect soil carbon stocks, this is a substantial limitation to interpretation of trends in and causes of stock change.

There is recent evidence (NSD deep profile re-sampled sites) to suggest that soil carbon stocks are not at steady-state for some soil-climate-management combinations. In the case of dairy pastures on lowland non-allophanic soils, soil carbon stocks appear to have declined in the top 30 cm over the last 20 years. However, it is not yet possible to say if these losses have occurred recently, have been occurring gradually over time, or have a wide geographic spread. There is also little understanding of the processes controlling, and factors contributing to, such losses.

By contrast, on dry stock and hill country pastures it appears that soil carbon stocks have increased slightly over the last 20 years. Again, there is little confidence in predicting future trends in soil carbon stocks in these soils, given the lack of causal understanding. The division between losses in lowland soils and gains in hill land soils might also be misleading, and the division between lowland and hill-land is largely arbitrary. The observed difference in behaviour might be reflecting a combination of differences, including contrasting pasture development histories, management intensity, fertiliser applications rates, and the degree of stability or disturbance (e.g., erosion; mechanical impacts of sheep versus cattle on soil structure and other properties). Grouping soils based on land form and/or land use classes may therefore have limited utility.

To progress development of carbon stock databases, and knowledge and understanding of the dynamics of C cycling in our pastoral agricultural soils, the following recommendations are made.

Addressing gaps in databases by selective sampling

- (i) Identify, prioritise and sample soil to a depth of at least 30 cm, from pasture sites for which there are long-term, well-documented management practices. As part of the programme of work, explore the relationships that would enable extrapolation of soil sample data from shallow depths to 30 cm depth, to make better use of the wide range of datasets with topsoil data. This would also need to identify whether or when such extrapolation would be valid.
- (ii) Obtain measurements of changes in soil carbon stocks with time for different land uses (both management type and intensity), land form, soil order and climate combinations to identify combinations that are at steady state, and losing or gaining carbon. This information is critical to:
 - refine the extent and national importance of the issue;
 - inform and focus the development of testable hypothesis for processes controlling observed changes in soil carbon stocks. This work should make use of existing long-term trial sites whenever possible.

Modelling soil C changes

Further soil carbon modelling capacity needs to be developed in New Zealand to facilitate process-based extrapolation through time and in space of measured datasets of soil carbon stocks and change. This would:

- improve understanding of likely long-term consequences of management practices for soil C, as these are difficult to determine them from short- to medium-term measurements alone;
- allow exploration of likely process-based explanations for the changes in soil carbon stocks observed in existing datasets, and hence to generate testable hypotheses for carbon-stock changes;
- provide an ability to assess the effect of current and future management practices on long-term soil carbon stocks; and
- improve understanding of temporal and spatial variability in soil carbon stocks and change, and facilitate better predictions of regional- to national-scale soil carbon stocks.

Specific data collection programmes

The more general gap-filling data collection programme referred to above also needs to be supplemented by a programme of work targeted at the following specific requirements:

- Experiments designed to test process-based hypotheses for changes in soil carbon stocks with land management practice.
- Sampling programmes to provide an understanding of processes leading to soil carbon change as a function of management type, intensity, soil type and climate, and that will facilitate the development and validation of process-based models.
- Experiments designed to test mitigation approaches for firstly maintaining and then enhancing soil carbon stocks especially in dairying and other intensive land

management systems, and also for further enhancing the carbon sequestration apparently occurring in North Island hill country soils.

5.9.3 Implications of accounting and mitigation options for New Zealand's post-2012 net position

At present, no defensible advice can be given on mitigation options for soil carbon stocks under pastoral agricultural management. It appears, however, there may be risks of soil carbon loss associated with:

- intensive dairying on some soil types, perhaps as a function of the intensity of management practices and varying with climate. This would be consistent with lowered below-ground inputs due to reduced production levels under high livestock densities.
- soil cultivation as part of forage cropping to support pastoral agricultural systems. However, such losses are likely to be recovered provided the cropping is temporary and the area is returned to grassland;
- cut-and-carry systems, which have recently begun to be considered for high-end production systems. Such systems could be expected to reduce inputs to pastoral systems and may result in at least an initial reduction in soil carbon stocks.

6. Effects of Cropping on Soil Carbon

Denis Curtin (Crop and Food), Mike Beare (Crop and Food)

6.1 Introduction

Compared with pastoral agriculture, cropping has usually been considered a less important land use in New Zealand, although it is more strongly represented in some regions (e.g., Canterbury) than in others. Nonetheless, approximately 240 kha are currently classed as cropping land under the Agricultural Production Census (2007), and carbon losses associated with cropping can be substantial. The area of cropping land is thought to be changing only slowly due to most agronomic soils already being utilised, although the drive for increased dairy production may be resulting in an increase in use of forage crops—the area of which presently remains unquantified.

There is a large body of international work on soil carbon losses under cropping, although translation to New Zealand conditions is not necessarily straightforward, given our temperate, moist climate and soil types. In particular, our soils tend to have high carbon contents by international standards, which might suggest the potential for greater losses. However, New Zealand's crop productivity is also much higher than international averages and the increased inputs to the soil from crop roots and above-ground residues are a definite and probably strongly compensating factor.

Changes in soil carbon stocks with cropping management are often reasonably slow (although generally both faster and larger than those experienced under pastoral agricultural management). This, coupled with the usual soil heterogeneity, generally requires that long-term trials be conducted to detect significant change, and with this has come problems of maintaining funding for data collection programmes. Nonetheless, several long-term programmes exist in New Zealand (as reviewed below); although to retain their value for soil carbon will require significant investment in some cases.

Like soil carbon change under pastoral agriculture, changes in soil carbon on cropping lands is already reported under the UNFCCC, and may need to be accounted as part of future commitments under the Kyoto Protocol. If in the future Article 3.4 of the Protocol were to become mandatory, or if any form of wider net-net accounting were introduced, accounting of soil carbon change in cropping soils would become compulsory.

In this Section of the report we review the results of New Zealand and international studies on the effect of cropping on soil carbon stocks. Soil carbon data for New Zealand cropping soils have been obtained from published reports, and from several unpublished data sets. The datasets are described in the Sections that follow, and their limitations assessed from a carbon accounting perspective. Estimates of current soil carbon stocks (0–30 cm) in cropland are made using current best-available data for soil carbon stocks and change, and cropland (land used for production of arable and vegetable crops) area statistics obtained from the 2007 Agricultural Production Census. Management options to increase carbon sequestration in croplands are evaluated based on New Zealand and international literature.

6.1.1 Datasets reviewed to determine the effects of cropping practice

The datasets reviewed in this part of the report comprise the five major New Zealand soil datasets that include measurements of soil carbon under cropping. All these datasets are under the custodianship of the CRI Crop and Food Research. A number of other New Zealand-cropping related studies and datasets are also reviewed here, although more briefly than the major datasets. The datasets and studies are the:

- Land Management Index (LMI) dataset
- ECAN A&P dataset
- Land-Use Change and Intensification (LUCI) dataset
- Soil Quality Management (SQMS) dataset
- Millenium Tillage Trial (MTT) dataset.

More briefly reviewed are:

- Studies of the effect of organic arable cropping systems on soil carbon
- Studies of the effect of cropland restoration on soil carbon
- Other New Zealand studies of arable cropping impacts on soil carbon.

Information about the five major datasets listed above that provide measurements of soil carbon under croplands is given in some detail in Appendix 2 of this Chapter, in the same standard format used for summarising the pastoral agricultural soil carbon datasets. The information provided includes site location, land use and management, sampling depths, methods of carbon measurement, and other biophysical attributes. Any other available land management data that could be used to assist interpretation of data trends have also been given in the Appendix.

6.2 Review of New Zealand Datasets

6.2.1 Land Management Index dataset

Dataset summary

The Land Management Index (LMI) dataset was collected for the purpose of developing a decision support system that farmers and land managers can use to (1) track changes in soil quality and predict risks to productivity losses or gains based on current management, and (2) to predict the effects of a change in management on soil quality and productivity before applying the change to the paddock.

The dataset comprises soil quality indicator measurements from 746 paddocks sampled between July 2002 and July 2007 as part of the LMI project. The paddocks represent seven land uses (mixed and intensive arable and vegetable cropping, dairy pasture, intensive bull/beef pasture, and extensive sheep/beef pasture) spread across seven different New

Zealand regions (Canterbury, Southland, Auckland, Waikato, Hawke's Bay, Manawatu and Gisborne). The paddocks sampled are located on key soil types representative of the major agricultural land uses in each region. The soil carbon (C) data were collected from 0–15 and 15–30 cm sample depths and are accompanied by bulk density measurements at these same depths.

The LMI soil quality dataset is closely aligned to comprehensive soil and crop management history information that is held in Crop and Food Research's Soil and Land Management Database. The database contains detailed information on the management practices used to establish and manage the crops and pastures grown during the 10 years preceding the measurement of LMI indicators—information on tillage types and frequency, irrigation, fertiliser, crop residue management, and grazing practices. The primary contacts for the data are Mike Beare and Erin Lawrence at Crop and Food Research, Lincoln.

The LMI dataset was not collected for the explicit purpose of soil carbon accounting. However, the dataset represents what is probably the largest and most comprehensive dataset suitable for quantifying soil C stocks under the major agricultural land uses in New Zealand. The dataset can also be used to quantify the magnitude of soil C stock change under land-use change, and the impacts of specific management factors (e.g., irrigation, tillage, winter cover crops) on C stocks in key land uses could be evaluated. The LMI dataset does not include data for many horticultural land uses (e.g., pipfruit, kiwifruit, viticulture, etc.), nor data for forestry and hill and high country pastoral farming systems. Some soil orders and several regions (e.g., Gisbone, Manawatu) are also under-represented in the dataset. The dataset also lacks C stock data for several other regions in New Zealand (e.g., Northland, Taranaki, Wanganui), although all of the major cropping regions are represented.

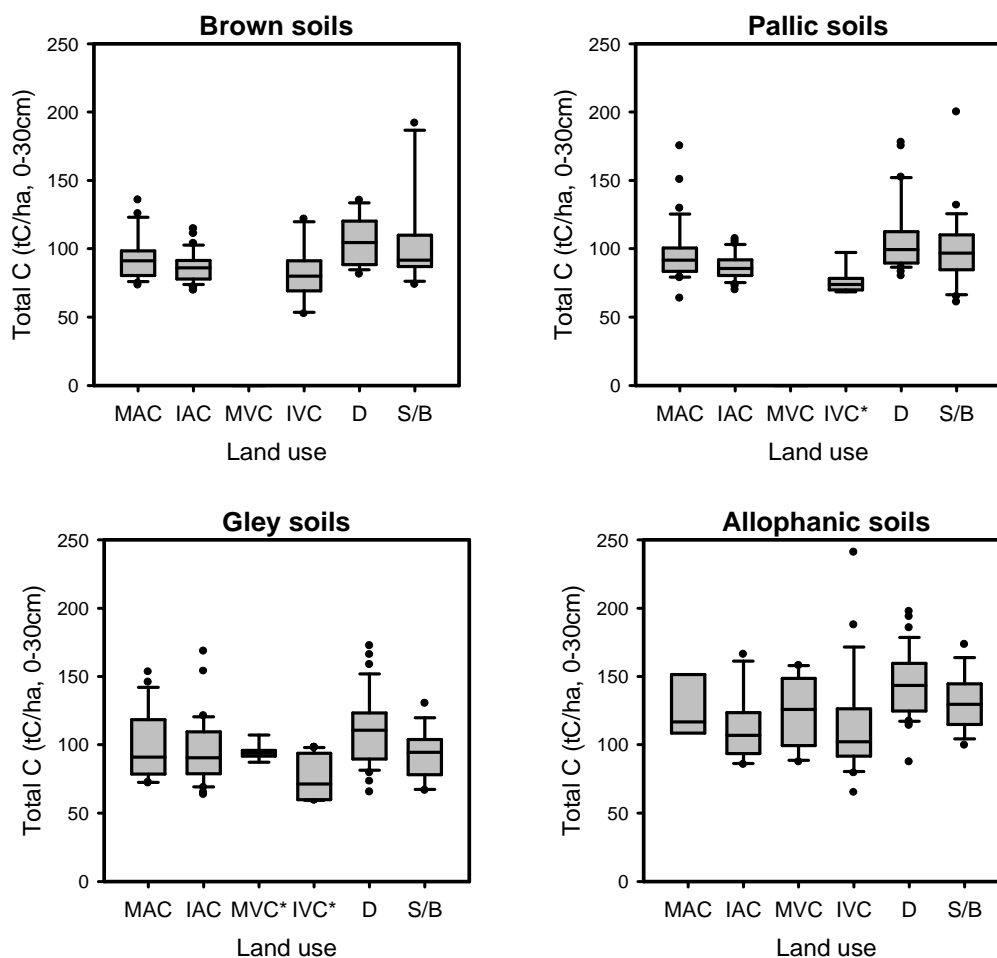
Main findings relevant to change in soil carbon stocks

For the purposes of this report, a preliminary analysis of the LMI dataset has been completed within the confines of the available time and funding.

The individual soil C concentrations and bulk density measurements were used to calculate soil C stocks (t C ha^{-1}) in the top 30 cm of soil at each sample location in each sampled paddock. The paddock average values were then used to calculate the average C stocks for each land use by soil order combination. A detailed analysis of land-use effects for all soil orders represented in the dataset and the individual soil types is beyond the scope of this project. The data presented in Figure 4 are preliminary only, and require closer scrutiny before being adopted as part of policy applications. The key points are:

- (i) Median C stocks tend to be highest under dairy pasture and lowest under intensive vegetable cropping regardless of soil order
- (ii) In general, Brown and Pallic soils tend to have a narrower range of values than Gley and Allophanic soils
- (iii) The effects of different cropping land uses on soil C stocks relative to sheep/beef pasture tend to be greater for Allophanic and Gley soils than for Brown and Pallic soils.

Figure 4 Box and whisker plots showing the distribution of soil C stocks representing each of the major land uses sampled under four major soil orders (Brown, Pallic, Gley and Allophanic soils). The land uses plotted are: mixed arable cropping (MAC), intensive arable cropping (IAC), mixed vegetable cropping (MVC), intensive vegetable cropping (IVC), dairy (D), and extensive sheep/beef pasture (S/B). Each box represents the middle 50% of the values measured for each land use, the line across each box is the median value and the values plotted outside the boxes are the upper and lower quartiles of values.



Future plans

The LMI dataset was not collected for the explicit purpose of soil carbon accounting. However, it is probably the largest and most comprehensive dataset suitable for quantifying soil C stocks under the major agricultural land uses in New Zealand. The dataset can also be used to quantify the magnitude of soil C stock change under land use change and the impacts of specific management factors (e.g. irrigation, tillage, winter cover crops) on C stocks in key land uses. Preliminary findings from this dataset are described above. A much more comprehensive analysis of the data set could provide valuable information on:

- (i) the mean, median and statistical range of soil C stocks in the top 30 cm of the soil profile under the major agricultural land uses, by soil order and dominant soil types;

- (ii) the magnitude of C stock change under different agricultural land uses relative to extensive sheep pasture;
- (iii) the effects of key management factors under different cropping (mixed and intensive arable and vegetable cropping practices in various regions) and pastoral land uses (Dairy, extensive sheep and intensive bull/beef farming in key regions) on soil C stock change;
- (iv) the management systems that offer the greatest potential of mitigating C losses.

6.2.2 ECan A&P dataset

Dataset summary

In 1999, Environment Canterbury (ECan) and Crop & Food Research (CFR) initiated a long-term soil quality monitoring programme to obtain information on soil conditions for different land-use/soil-type combinations. The Arable and Pastoral Monitoring dataset forms part of ECan's state of the environment monitoring and reporting programme.

As of June 2007, the Arable and Pastoral Monitoring dataset comprised soil quality indicator data from 220 paddocks sampled between 1999 and 2006. The paddocks sampled represent three broad land use categories (i.e. long-term pasture, short term pasture or arable and long-term arable cropping) on each of 12 different soil types commonly found on the Canterbury plains and downs. From 2008 onward, previously sampled sites are scheduled for resampling (9 years post original sampling) in order to provide information on changes in soil quality over time. The soil C data collected before June 2006 was based on 0–15-cm samples only, while data collected since that time and in the future will be based on the top 30 cm (0–15 and 15–30 cm) of the soil profile. The soil C data are accompanied by bulk density measurements and a number of other soil chemical, physical and biological measurements. The primary contacts for the data are Mike Beare and Craig Tregurtha at Crop and Food Research, Lincoln.

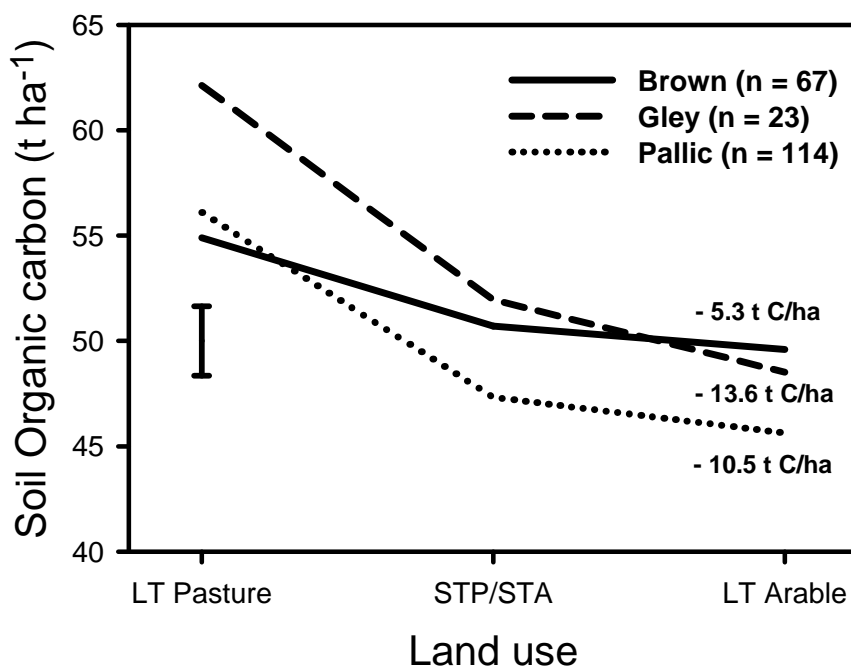
The Arable and Pastoral Monitoring dataset was not collected for the explicit purpose of soil carbon accounting. The current dataset has limited application for C accounting owing to the shallow depth (0–15 cm) of sampling. However, the existing data do provide a relative measure of the resistance of different soil types to C loss following conversion from long-term pasture to continuous cropping land uses. From 2008 onwards, the C stock measurements made to 30 cm will have more direct applications to soil C accounting and quantifying the change in soil C stocks from long-term pasture to long-term continuous cropping land uses on different soil types.

Main findings relevant to change in soil carbon stocks

For the purposes of this report, a preliminary analysis of the Arable and Pastoral monitoring dataset has been completed within the confines of the available time and funding. The individual soil C concentrations and bulk density measurements were used to calculate soil C stocks (t C ha^{-1}) in the top 15 cm of soil at each sample location in each sampled paddock. The paddock average values were then used to calculate average C stocks for each land use by soil order combination. A detailed analysis of the land use effects for the individual soil types is beyond the scope of this project but could be included in future MAF funded soil C stocks and change research.

The results showed that the magnitude of soil C stock change from long-term pasture to long term arable cropping differs significantly between soil orders (Fig. 5). In general, the lighter, well-drained Brown soils had the lowest C stocks under longer term pasture (LT Pasture) and smallest loss of soil C under long-term continuous cropping (LT Arable) relative to long-term pasture. In contrast, the heavier, usually poorly drained Gley soils tended to maintain the highest C stocks under long-term pasture and showed the greatest losses of C under long-term continuous cropping. Soil C stocks under cropping on Pallic soils (mostly imperfectly drained) tended to be lower than those of the Brown and Gley soils, though the losses of C relative to long-term pasture were intermediate to those of the Brown and Gley soils.

Figure 5 Average soil C stocks (t C ha^{-1}) in the top 15 cm of Brown, Gley and Pallic soils under long-term sheep pasture (LT Pasture), short-term pasture of arable cropping (STP/STA), and long-term continuous arable cropping (LT Arable). The difference in C stocks between LT Pasture and LT Arable is also shown as the average C loss under continuous cropping, n = the number of paddocks representing each soil order in the data set.



Future plans

ECan is committed to repeat sampling of each paddock in the programme (on a 9-year-return cycle). With suitable funding, the C stock measurements made from 2008 onward (top 30 cm) will have useful applications to soil C accounting and quantifying the change in soil C stocks from continuous pasture to continuous cropping land uses on different soil types.

6.2.3 Land Use Change and Intensification (LUCI) – Canterbury

Dataset summary

The goal of the FRST funded Land Use Change and Intensification (LUCI) programme (2003–2008) is to provide integrated knowledge and tools required by land users and policy makers to assess the environmental impacts associated with land-use change and intensification of agricultural practices. One key focus of this programme has been to

quantify the extent and rate of change in soil quality and plant production following changes in land use under typical management practices in Canterbury.

The extent and rate of soil quality change is being quantified under several important forms of land-use change (LUC) on the Canterbury plains. This is based on replicated paddocks undergoing conversion from extensive dryland sheep pasture or dryland mixed cropping to intensive irrigated cropping on soil types representing well drained, imperfectly drained and poorly drained soils. The dataset includes annual measurements of soil physical, chemical and biological properties on paddocks undergoing the first 4–5 years of land-use change and on other paddocks representing longer periods of intensive irrigated cropping (up to 20 yrs) on the same soil types. The soil types included in this analysis of land use change impacts were selected to represent well drained, imperfectly drained and poorly drained soils common to the Canterbury Plains. As of June 2007, a total of 87 paddocks had been sampled, with many of these paddocks involved in repeated measurement of the soil quality indicators. The soil C data were collected from 0–15 and 15–30 cm sample depths and are accompanied by bulk density measurements at these same depths. The primary contacts for the data are Mike Beare and Erin Lawrence, at Crop and Food Research, Lincoln.

The LUC dataset was not collected for the explicit purpose of soil carbon accounting. However, it represents one of only a few datasets suitable for quantifying the actual rate of soil C change under land use change in New Zealand. The dataset allows quantification of soil C stocks (t C ha^{-1} , 0–30 cm) under major agricultural land uses on the Canterbury and the impacts of specific management factors (e.g., irrigation, tillage, winter cover crops) on C stocks in key land uses. The LUC soil quality dataset is closely aligned to comprehensive soil and crop management history information that is held in Crop & Food Research's Soil and Land Management Database. The database contains detailed information on the management practices used to establish and manage the crops and pastures (tillage types and frequency, irrigation, fertiliser, crop residue management, grazing practices) grown during the 10 years preceding the soil quality measurements.

The dataset allows quantification of C change, both the rate and extent of C change with change in management. Collection of the LUC soil quality dataset was scheduled to be completed in June 2008. Detailed analysis of the data were not yet available at the time of writing this report.

Closely aligned and complimentary data on the rate of soil quality (including soil C) change during the conversion of extensive dryland sheep pasture or dryland mixed cropping to intensive irrigated dairy farming is also being collected for well drained soils in Canterbury under the Sustainable Land Use Research Initiative (SLURI).

Main findings relevant to change in soil C stocks

The collection of this dataset has only recently been completed and it awaits detailed analysis to determine the main findings.

Future plans

Analysis and interpretation of the data will focus on quantifying the extent and rate of change in soil quality and plant production following changes in land use under typical management practices in Canterbury. Although the data set was not explicitly collected for purposes of soil C accounting, this objective could be included as part of the data analyses planned for

2008/09. Some supplementary funding would be required to assist with this analysis and the data interpretations.

6.2.4 Soil Quality Management System (SQMS) dataset

Dataset summary

This SQMS dataset was collected as part of the development of the Soil Quality Management System (SQMS). The SQMS was developed to assist farmers with on-farm monitoring of soil quality to improve soil management decisions. Soil quality indicator data (including soil C measurements) were collected from mixed and intensive cropping and extensive sheep paddocks representing some of the most common cropping soils in the Canterbury and Southland regions.

The Canterbury dataset includes annual measurements of soil quality on 69 paddocks between 1999 and 2001. The Southland paddocks (31) were each sampled annually between 2002 and 2004. The soil C data are based on composite samples made up of 15 soil cores collected along a W or Z transect in each paddock. The soil C data are accompanied by bulk density measurements and a number of other soil chemical, physical and biological measurements. Soil and crop management history information (crop type, tillage type, irrigation, residue management, etc.) was also collected for the 10 years preceding the soil quality assessments in each paddock. The primary contacts for the data are Mike Beare and Craig Tregurtha at Crop and Food Research, Lincoln.

The SQMS dataset was not collected for the explicit purpose of soil C accounting. The dataset has limited application for this purpose owing to the shallow depth (0–15 cm) of sampling. However, the existing data do provide a measure of the range of topsoil C stocks for extensive sheep and cropping land uses in Canterbury. When combined with the management history information, the dataset has also proved valuable in evaluating the effects of key management factors (e.g., tillage type and intensity, residue management practices, crop rotations etc) on topsoil quality. A further analysis of the dataset is needed to quantify the impacts of management practices on soil C stocks.

Main findings relevant to change in soil C stocks

The dataset has not been adequately analysed with respect to soil C stocks and change.

Future plans

Parts of the SQMS soil quality dataset are currently being written up for publication in international journal articles. Analysis of the top soil C stocks data in relation to management history may be included in this analysis.

6.2.5 Millennium Tillage Trial (MTT) dataset

Dataset summary

The Millennium Tillage Trial (MTT) was established to identify tillage and cover crop management practices that maintain organic matter levels, reduce structural degradation, increase nutrient use efficiency and minimize nutrient losses in order to sustain arable cropping out of an improved condition under long-term grass pasture. The MTT is part of the FRST funded Land Use Change and Intensification (LUCI) programme (2003–2008) and contributes to the development of best management practices for the intensification of arable cropping. However, the dataset also has the potential to contribute to our understanding of

the short-term soil quality changes (including change in C stocks) that occurs during establishment and maintenance of break crops in pastoral re-grassing (grass renewal) rotations (Beare et al. 2008).

The trial is composed of six tillage treatments (based on different combinations of spring and autumn tillage) plus a 'control' of uncultivated permanent pasture alongside a permanent fallow treatment. The three main tillage methods used were Intensive tillage, minimum tillage and no-tillage. Each of the main plots has been split to compare winter cover crops to no cover crop (winter fallow) treatments. Each treatment is replicated three times, giving a total of 42 treatment plots. Details on the trial design, including the specific management practices used can be found in the Cropping Section.

The MTT dataset includes annual measurements of a wide range of soil physical, chemical and biological properties from the treatment plots on Lincoln-based trial site. The soil at this site is Wakanui silt loam that had been under at least 15 years of continuous sheep pasture management prior to establishing the trial. The soil C measurements have been made annually from composite samples made up seven soil cores (each 7.2 cm in diameter) collected from 0–7.5, 7.5–15, 15–25 and 25–30 cm sample depths in each plot. The soil coring equipment was designed to collect relatively large samples suitable for bulk density analyses and a wide range of other physical, chemical, biological measurements at each depth. Considerable care has been taken to complete very precise accounting of soil C and N stocks under the different treatments over time using the equivalent mass sampling method (Ellert et al. 2001). The primary contacts for the data are Mike Beare, Denis Curtin and Trish Fraser at Crop and Food Research, Lincoln.

The MTT dataset was not collected for the explicit purpose of soil carbon accounting. However, it represents one of only a very few datasets that are suitable for quantifying the extent and rate of soil C change during the conversion of pasture to arable cropping in New Zealand. It also represents the only long-term replicated trial whereby the effects of different tillage practices (including no-tillage) on soil C stocks can be evaluated. The dataset currently contains annual measurements of soil C stocks (t C ha^{-1} , 0–30 cm) under each of six different tillage systems (ranging from no-tillage to continuous intensive tillage) in an arable cropping rotation that was established out of grass pasture and has been under continuous treatment for 8 years. We are currently reviewing the trial to determine if it should be maintained in the longer term.

Main findings relevant to change in soil C stocks

In the short term (1–2 years), C loss was more rapid under intensive cultivation. In longer term (7–8 years), tillage effects on soil C were relatively small.

Future plans

The trial is being modified to accommodate comparison of spring and autumn sown crops (currently main crops are sown in spring, whereas most farmers sow cereals in autumn).

6.2.6 Straw field trial dataset

Dataset summary

As post-harvest crop residues represent a major input of C to arable cropping systems, it was hypothesised that the way in which these residues are managed may have a significant influence on soil C stocks. A 6-year study (1992–98) in Canterbury (Lincoln) determined the

effects of three straw-management practices [(1) straw incorporated; (2) straw baled and removed; and (3) straw burned] on soil C (Curtin & Fraser 2003).

In the straw-incorporated treatment, about 25 t/ha of straw (~11 t C/ha) was returned to the soil during the course of the trial. However, there was no significant effect ($P > 0.05$) of straw management on total soil C (0–15 cm). Measurements of straw decomposition using the litter bag technique (carried out in association with the trial) indicated that much of the incorporated straw would have decomposed and the small fraction of straw-C retained in the soil (estimated at 2–3 t C/ha) would have been difficult to detect against a background soil C content of over 50 t/ha in the top 15 cm.

Main findings relevant to change in soil C stocks

The duration of the trial was not sufficiently long. Estimates of C gains due to crop residue retention range from 0.1 t C/ha/year (Sampson et al. 2000) to 0.7 t C/ha/year in some European studies (Smith et al. 2005). Trials would need to run over 15–20 years to detect such small annual gains in soil C.

Future plans

None.

6.3 Review of Other New Zealand Studies

The studies briefly reviewed in this section vary substantially in usefulness, being generally characterised by shallow sampling depths and, in a number of cases, by lack of bulk density measurements. Also, some datasets have yet to be analysed, and some are considered of limited value because of experimental inadequacies. Consequently, although the studies are viewed for the sake of completeness, most are dealt with here in a fairly cursory manner. More details on all studies mentioned in this Section can be found in Appendix 3 of this Chapter.

6.3.1 Organics datasets

Formal trials with the objective of providing direct comparison of C stocks under organic and conventional arable cropping have not been conducted in New Zealand. However, a number of published studies are available in which paired comparisons were made between commercial organic farms and nearby conventional farms (Reganold et al. 1993; Nguyen et al. 1995; Murata & Goh 1997). These studies have been reviewed by Condrón et al. (2000), who concluded that soil organic matter concentration is generally higher under organic versus conventional production systems. However, the available data are poorly suited to assess whether accountable carbon stocks (i.e. those to 30 cm) are also higher under organic than conventional farming because:

- a comparison between conventional and biodynamic farming, for a wide range of production systems, has been reported by Reganold et al. (1993). Soil sampling was carried out to only 10 cm. Carbon concentration was higher under biodynamic than conventional farming practices, but bulk density was lower, resulting in no significant difference in soil carbon stocks.

- a comparison between biodynamic and conventional mixed cropping systems has been reported by Murata and Goh (1997). Soil sampling was carried out to 15 cm. No bulk density data were obtained, and so carbon stocks could not be determined.
- A comparison between biodynamic/organic and conventional mixed cropping systems (in both the crop and interspersed pasture phases) has been reported by Nguyen et al. (1995). Soil sampling was carried out to only 7.5 cm. No bulk density data were obtained, and so carbon stocks could not be determined.

There is also one major unpublished study, which may be more useful than the above studies for comparing soil carbon stocks under organic and conventional cropping: the Kowhai Farm dataset. Kowhai farm was established as a joint venture between Lincoln University and Heinz-Watties in spring 1999 to demonstrate the economic viability and environmental sustainability of farm-scale certified organic production. This demonstration farm is composed of 6 paddocks, most of which are represented by Wakanui, Templeton and Paparua silt loam soils. The dataset was collected to describe changes in soil quality during the conversion to certified organic production, but could be used to determine carbon stocks to 15 cm depth. Measurements are available prior to initiating the organic farm conversion, and on other adjoining conventional cropping paddocks before and during conversion to fully certified organic production.

Soil and crop management history information (crop type, tillage type, irrigation, residue management, etc.) was also collected for a period of about 10 years preceding the measurements. To date, however, no suitable analysis for determining soil carbon stocks has been completed.

6.3.2 Restorative crops trial dataset

A 6-year (1989–95) experiment was carried out by Francis et al. (1999) to evaluate the ability of a variety of crops to improve the fertility and physical condition of an intensively cropped, degraded soil (Wakanui silt loam) in Canterbury (Lincoln). Treatments included perennial pastures, annual pastures, and arable crops. Soil C was determined after 3 and 6 years, but treatment effects were generally not significant until the sixth year. At that time, significant differences of about 10 t C ha^{-1} were evident between uncultivated and annually cultivated treatments—including annual re-establishment of pasture by conventional tillage. Carbon stocks under annual direct-drill pasture re-establishment were effectively the same as under other uncultivated treatments.

6.3.3 Cropland data from other datasets

A range of other New Zealand studies have developed datasets that include measurement of soil carbon, although none have had the explicit objective of determining soil carbon stocks. These datasets are of varying usefulness, and comprise:

- Data on the effect of land use on soil quality, including C stocks, for the Auckland, Waikato and Canterbury regions. Data are limited to the top 10 cm only. These data have been reported by Schipper and Sparling (2000), and Sparling and Schipper (2004). The datasets show mean soil carbon stocks under cropping are about 20 t C ha^{-1} less than under pastoral agricultural farming, with a slightly smaller mean difference between cropping and indigenous forest soils.

- The 500 Soils Dataset: a programme of sampling to determine the impact of land use on the properties of a large number of soils from the predominant New Zealand intensive agronomic land uses (dairy pasture, sheep and beef pasture, cropping and horticulture, plantation forestry and indigenous vegetation). The dataset encompasses all the major soil Orders across New Zealand, and follows strict sampling protocols. Data are limited to the top 10 cm only, and this dataset has yet to be analysed for estimation of soil carbon stocks.
- Canterbury tillage trial dataset. This work has been reported in Francis and Knight (1993), and compared the effects of conventional cultivation (ploughing to 18–20 cm) and no-tillage at two sites in Canterbury. However, sampling was to 15 cm depth only, whereas cultivation was to 18–30 cm depth. It is therefore unlikely any valid conclusions can be drawn from this dataset.
- Ohakea silt loam tillage trial dataset. This work has been reported in Aslam et al. (1999, 2000), and compared the effect of plough cultivation and no-tillage treatments on soil organic matter following conversion of a permanent pasture to arable cropping. Sampling was to 10 cm only, and only for 2 years. It is therefore unlikely that any valid conclusions can be drawn from this dataset.

More useful in general than the above datasets are data reported in the study of Shepherd et al. (2001), on the impact of arable cropping on former pasture lands. Soil properties measured included soil carbon stocks, and the study included measurements of the change in soil carbon stocks over time. This work is unique in New Zealand in this respect, and although measurements were only to 20 cm this at least was consistent with the depth over which most changes were expected—that is, it covered the depth over which cultivation occurred. Details of this study can be found in Appendix 3 of this Chapter. In summary, the study showed that:

- the magnitude and rate of soil carbon losses after conversion to cropping depended on the soil and crop type. Similarly, gains possible on re-conversion to pasture also depended on soil type, at least, with some types showing very limited gains and others gaining as much as $1.1 \text{ t C ha}^{-1} \text{ yr}^{-1}$.
- Average medium- to long-term losses (7–40 years, depending on the site) as a result of continuous cropping ranged from about 1 to $1.5 \text{ t C ha}^{-1} \text{ yr}^{-1}$.
- The smallest losses observed averaged $0.2 \text{ t C ha}^{-1} \text{ yr}^{-1}$ over 20 years for barley cropping on an Egmont silt loam, and the largest averaged $1.7 \text{ t C ha}^{-1} \text{ yr}^{-1}$ over 20 years on a fine-textured Kairanga soil.

6.4 Datasets for Determining Cropping Land Soil Carbon: A Summary

Datasets containing information on cropland soil carbon stocks have not been collected in New Zealand for the explicit purpose of carbon accounting. Most datasets therefore have not sampled to 30 cm, with depth of 7.5–15 cm being most common. However, there are three datasets that do have a 30 cm sampling depth, and these can provide initial estimates of carbon stocks under cropland. With further analysis and continued sampling, these data sets

can also provide information on the rates of change in stocks. Crop and Food Research are the custodians of these datasets, which are:

- The LMI dataset: a very comprehensive dataset in terms of both regional coverage and the number of soil types sampled. The dataset has information not only on cropland soil carbon stocks, but also on stocks under dairying and drystock farming in the same soil/climate categories. Extending the analysis of the LMI dataset to support carbon accounting should therefore be given priority, given its wide coverage and relatively large number of samples. Moreover, if the LMI sampling programme can be continued, it will allow determination of rates of change in soil carbon stocks for at least 3 key land uses. This again emphasises that securing and expanding future funding for the LMI dataset is a high priority.
- The LUCI dataset: this dataset can potentially provide valuable initial information on several important forms of land-use change, albeit only for soils on the Canterbury plains. However, this is a key region in terms of both land conversion to cropping, and management intensification. The dataset includes irrigation treatments, which are topical. Even more important, time sequential data are available with sufficient temporal frequency potentially to develop both rates of change and the shape of the time dependence of soil carbon stocks following land conversion/intensification. Analyses of data in this dataset have not yet been completed. Adding a specific soil carbon focus to this analysis is therefore a high priority for future funding. As this dataset also comprises time dependent data, elevating the LUCI programme to the status of a long-term study is also a high priority.
- The Millennium Tillage Trial dataset: this dataset includes information on soil carbon stocks from the only long-term trial in New Zealand of the effects of different tillage practices—a key issue in terms of decisions about use of no-/low-till practices for mitigating soil carbon losses under cultivation. Initial measurements need to be analysed so that a decision can be made as to the longer term future of this potentially very important study.

Probably the next most useful information on the effects of cropping comes from the work reported in Shepherd et al. (2001), where differences in soil carbon stocks between pasture and adjacent cropland areas were reported to depths of 20 cm. Although sampling was not to 30 cm depth, it did at least cover the cultivated depth, and very importantly provides time sequential data allowing useful information on both the magnitude and rate of carbon stocks to be estimated. Data are available for a variety of soils, regions and crops. Further analysis of the data reported by Shepherd et al. (2001), focussed on soil carbon issues, may be warranted.

6.5 Review of Key International Information

New Zealand's arable cropping systems are highly productive by international standards. This can be attributed to favourable climatic conditions coupled with the use of irrigation where rainfall is inadequate. Climatic conditions have a strong influence on C inputs (above-ground crop residues and roots) and on the decomposition rate of both fresh residues and native soil organic matter. Given these differences in soil and climatic conditions, international findings are not necessarily directly transferable to New Zealand. However,

information from countries with comparable climates can be useful in filling knowledge gaps, particularly where there is a dearth of local, long-term data.

The effects of crop residue management on soil C stocks require long-term assessment because the annual change in soil C associated with a given practice is usually small in relation to background variability in C stocks. As discussed above, contrasting straw management practices (removal vs retention) did not have a detectable effect on soil C within 6 years in Canterbury (Curtin & Fraser 2003). Much longer studies of straw management effects have been conducted in North America [e.g., a 30-year study by Campbell et al. (1991) in Canada] and Europe (e.g., 35 year study in Uppsala, Sweden; Persson & Mattsson 1988). European studies, which are probably the most relevant to New Zealand, have been analysed by Smith et al. (1997), who found that, when data from eight long-term experiments were pooled, soil C increased with rate of straw incorporation according to the equation:

$$y = 0.11 x + 0.19 \quad (1)$$

where y is the annual increase in soil C (%) and x is the amount of straw incorporated (t/ha per year). According to this equation, incorporation of 7 t/ha of straw (about the average straw production in Canterbury) would increase soil C by 1% annually. For an arable soil that contains about 70 t C/ha to 30 cm, this amounts to an annual C gain of 0.7 t/ha. This is also the suggested European value for the C sequestration potential of crop residues (Smith et al. 2005). This is much higher than the value of 0.1 t C/ha reported by Sampson et al. (2000). Although the European studies suggest that, over the long-term, straw retention may increase soil C, caution is needed in transferring the findings to New Zealand. Temperature has a major influence on the rate of straw decomposition (Douglas & Rickman 1992), and warmer conditions in New Zealand compared with European locations (e.g., annual degree days for Lincoln, Canterbury, are 4200 versus about 3400 for Rothamsted in the Southern UK) are likely to mean faster decomposition and less retention of straw-C in New Zealand soils.

The C sequestration potential of no-tillage has been studied extensively overseas. From a global database of 67 long-term experiments, West and Post (2002) estimated that a change from conventional to no-tillage may increase soil C by 0.57 ± 0.14 t ha⁻¹ per year. Increases in the rate of soil C sequestration following conversion to no-tillage were estimated to peak in 5–10 years, with soil C reaching a new equilibrium in 15–20 years. There is evidence that the C mitigation potential of no-tillage can differ regionally, depending on climate, crops grown, residue quality, and possibly other factors including earthworm activity (Gregorich et al. 2005). For example, cropping soils in semiarid western Canada often show increases in C after switching to no-tillage (Campbell et al. 1996), whereas in humid eastern Canada, no-tillage does not appear to have a beneficial effect on soil C (Gregorich et al. 2005). Recently, Blanco-Canqui and Lal (2008) questioned the view that no-tillage can increase soil C. They suggested that many studies reporting differences between conventional and no-tillage based on shallow sampling need to be re-evaluated. In studies in Ohio, Kentucky and Pennsylvania, soil C (0–10 cm depth) was higher under no-tillage than under conventional tillage in five of eleven comparisons, but total C to 60 cm was not significantly affected by tillage intensity (Blanco-Canqui & Lal 2008).

6.6 Mitigation Opportunities for Cropping Soils

6.6.1 Assessment of mitigation options

To increase soil C storage, it is necessary to either increase C inputs or decrease the rate of decomposition. Whereas arable soils in other countries may be amended with manures generated elsewhere, C inputs to New Zealand cropping soils consist of crop residues produced in situ. Managing crops to maximize yield (by providing adequate nutrients, especially N, and minimising moisture stress using irrigation) should also maximize C inputs in post-harvest residue. Crop type has a strong influence on amounts of C returned in plant residues, with perennial grasses (pasture) returning the largest amounts, vegetable crops the least, and cereals being intermediate. Yields of arable crops have increased over time due to improved cultivars and optimisation of management inputs, including irrigation. However, as burning continues to be a predominant residue management option, particularly in Canterbury, increases in residue amounts may not translate into increases in C in cropping soils. Currently, about 60 000 hectares of crop residues are burned annually (about 53 000 ha in Canterbury; 2007 Final Agricultural Production Statistics; www.stats.govt.nz).

As indicated above in Section 5.2.6, reliable local data for the C sequestration due to cereal residue retention are not available. If the European value of 0.7 t C/ ha/year (Smith et al. 2005) is accepted, elimination of straw burning could increase C sequestration in cropping soils by about 40 000 tonnes annually. Clearly, there is considerable uncertainty around this estimate and, given that crop residue decomposition is likely to be more rapid in New Zealand's warmer climate than in Europe, the C mitigation potential of crop residues may be far less than 40 000 t C/year.

Low disturbance tillage systems may be beneficial in increasing soil C in some soils, particularly where initial soil C stocks are very low (usually owing to long-term continuous cropping). Although part of the benefit may be due to improvements in yield and inputs of C in residues, the C sequestration potential of minimum- and no-tillage is generally attributed to a decrease in organic matter decomposition. Recently, there appears to have been a re-appraisal of the C sequestration role of no tillage with the realisation that inadequate sampling practices (particularly shallow, fixed-depth sampling) may have led to a bias in favour of no-tillage.

To carry out a proper comparison of C stocks under conventional- and no-tillage, it is essential (1) to include the entire cultivation layer and (2) ensure that the same mass of soil is sampled under the two tillage systems. Under intensive cultivation, C tends to be uniformly distributed by depth within the plough layer (usually the top 20 cm), whereas C may be concentrated near the surface in no-tillage. Several New Zealand studies have been biased in favour of no-tillage by the fact that shallow samples were used for comparisons. Differences in bulk density between tillage treatments (bulk density normally higher under no-tillage vs intensive cultivation) means that, in fixed depth sampling, the mass of soil sampled differs between treatments. To overcome this problem, a so-called equivalent mass method has been proposed whereby C stocks can be compared based on the same soil mass (Ellert et al. 2000). The dearth of reliable New Zealand data for C sequestration under no-tillage makes it difficult to estimate the change in soil C associated with expansion of the area of cropland that is direct drilled.

It is not known if irrigation expansion has affected soil C stocks in New Zealand cropland. Because straw is likely to be burned, particularly in Canterbury where most of the irrigated cropland is located, C inputs may not be greatly different under irrigated versus rainfed cropping. Also, use of irrigation can increase decomposition and C turnover. The net effect may be little change in C stocks. Under sheep-grazed pasture, border dyke irrigation has been shown to decrease soil C at Winchmore even though dry matter production and, by extension, C returns to the soil, were substantially greater than under dryland pasture (Condon et al. 2006).

6.6.2 Environmental co-benefits and risks

Elimination of straw burning

The main advantage of straw burning is the quick and convenient clearing of the land so that the next crop can be established without delay. It may also help in the control of certain pests and diseases so that more agri-chemicals may be needed if residues are not burned. In terms of greenhouse gas emissions, the burning of residues produces only about 10% more emissions (primarily from methane production) than does decomposition, and the emissions that would be avoided by not burning could be offset by increased emissions from machinery if greater use of agri-chemicals became necessary.

On the positive side, co-benefits of the elimination of stubble burning include improvements in air quality (reduction of smoke and particulates). Incorporation of residues can also temporarily immobilise N and may thus reduce the leaching of nitrate during winter.

Adoption of no-tillage

Co-benefits of the adoption of no-tillage are:

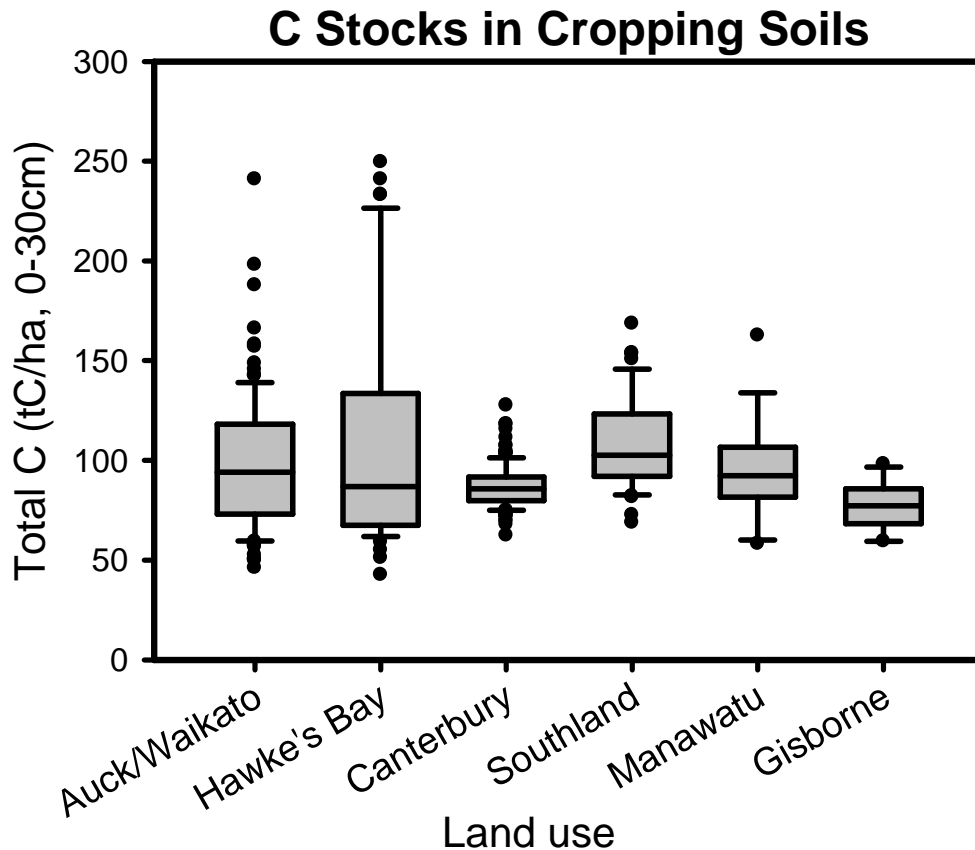
- (i) decreased fossil fuel consumption and a reduction in the associated CO₂ emissions;
- (ii) increased water use efficiency/greater water conservation; and
- (iii) reduction of soil erosion (and loss of carbon in eroded soil).

6.7 Current Best Estimates of Cropping Soil Carbon Stocks and Change

6.7.1 Carbon stocks and change

Estimates of soil C stocks to 30 cm were obtained from the LMI data set for land under arable and vegetable cropping in Canterbury (143 sites), Southland (49 sites), Auckland/Waikato (103 sites), Hawke's Bay (47 sites), Manawatu (16 sites), and Gisborne (18 sites). As discussed above, this is probably the largest and most comprehensive dataset suitable for quantifying soil C stocks under the major agricultural land uses in New Zealand. Under cropping, the median C level was highest (103 t/ha) in Southland (presumably reflecting heavier soil textures in that region) and lowest (77 t/ha) in Gisborne (Fig. 6). It should be noted that both Gisborne and Manawatu are relatively poorly represented in the data set (18 and 16 observations, respectively), compared with the other regions. The median and mean values for each region were very similar, except for Hawke's Bay where the mean exceeded the median by 20 t C/ha. In Hawke's Bay, and to a lesser extent in Manawatu, mean C stocks were strongly influenced by data from several sites with organic (peat) soils with very high C stocks.

Figure 6 Box and whisker plots showing soil C stocks by region, based on LMI dataset. Each box represents the middle 50% of the values measured for each land use, the line across each box is the median value and the values plotted outside the boxes are the upper and lower quartiles of values



The LMI dataset records current C stocks (samples were collected from 2002–07). Using median values for the seven regions and the areas of arable and vegetable crops in each region (obtained from the 2007 Agricultural Production Census), we estimated soil C storage by region (Table 19). This estimate of soil C storage assumes that arable and vegetable soils do not differ in C (further analysis of the data set will allow us to provide separate estimates for arable and vegetable soils). Several regions with small areas of cropland are not represented in the LMI data set. We assumed that median C levels in Northland and the Bay of Plenty were the same as in the Auckland/Waikato region and that Taranaki and Wellington soils had the same C level as those in the Manawatu. In the South Island, we do not have data for the Tasman/Nelson/Marlborough region or for Otago. Soil C in these regions was assumed to be that same as in Canterbury.

We estimated that the 238 000 ha of cropland in New Zealand contain about 21 Mt C (about 13.5 and 7.5 Mt C in the South and North Islands, respectively; Table 5.1). These estimates of current C storage in cropland should be regarded as a first approximation only, given that they depend on the validity of several assumptions. Tate et al. (2003) estimated 1990 cropland C stocks at 26 ± 3 Mt C. Their estimated cropland area (300 000 ha) was larger than our estimate (238 000 ha) because Tate et al. (2003) had included horticulture (90 000 ha).

With an adjustment for the differences in area, the estimated present-day cropland C stock value (21 Mt C) is not distinguishable from that for 1990 reported by Tate et al. (2003). Since the LUCI-derived estimate is independent of that used by Tate et al. (2003), this result is consistent with the view that no detectable changes in soil carbon stocks has likely occurred under cropland between 1990 and the present.

It is possible that soil C stocks in cropland may have been lower in 1990 than at present because:

(i) C inputs from plant residues were smaller (associated with lower crop yields) in 1990;

and

(ii) farmers used more intensive cultivation methods in 1990.

It is estimated that about 20% of all seeding is currently done by no-tillage, compared with < 4% in 1990 (C. Ross, Landcare Research, pers. comm.). Assuming that 0.57 t C/ ha/ year can be sequestered over a 10-year period following a switch to no-tillage (West & Post 2002), it can be estimated that the increasing adoption of no tillage might be have led to an increase in C stocks by about 0.2 Mt C since 1990. This represents a 1% change in C stocks, which would be too small to detect experimentally. Given that a large proportion of small grain residue is still burned, it is arguable that, when residue loss is countered by increases in crop production and no-till area achieved since 1990, total annual C inputs have not changed substantially over time. It may therefore not be unreasonable to conclude that C stocks in cropland present at 1990 are similar to those on that land at present.

Table 19 Cropland C storage estimated using soil C data from the LMI dataset and crop areas obtained from the 2007 Agricultural Production Census

Region	Total arable crops (ha)	Total vegetable crops (ha)	C stocks (t/ha)	Total stocks (all crops) (tonnes)
Northland	3162	1476	94	435 972
Auckland	2277	5292	94	711 486
Waikato	21 495	4027	94	2 399 068
Bay of Plenty	5349	83	94	510 608
Gisborne	3025	4907	77	610 764
Hawke's Bay	3856	8457	87	1 071 231
Taranaki	2490	18	92	230 736
Manawatu-Wanganui	9435	4524	92	1 284 228
Wellington	2976	46	92	278 024
North Island	55 613	22 650		7 532 117
Tasman	364	401	86	65 790
Nelson	0	0	86	0
Marlborough	1981	1454	86	295 410
Canterbury	122 707	12 528	86	11 630 210
Otago	9042	439	86	815 366
Southland	6133	74	103	639 321
South Island	142 267	15 481		13 446 097
Total New Zealand	197 881	47 170		20 978 214

Since 1990, there has been an increase in cropland area of about 30 000 ha. The IPCC default period for the transition time for soil carbon stock change following conversion of land to cropland is 20 years. The time sequential changes in soil carbon stocks following conversion of land to cropland under New Zealand conditions reported by Shepherd et al. (2001) is not inconsistent with such a transition time, and at present there seems little evidence for adopting a time constant other than the IPCC 20-year default value. Tate et al. (2003) reported a stock change factor of -10 ± 7 t C ha⁻¹ for cropland, which is within the error bounds of the mean difference in soil carbon stocks between pasture and cropland found in the LMI dataset study (c. 14 t C ha⁻¹), or the mean long-term change found following cropland conversion in the LMI dataset study (c. 10 t C ha⁻¹, albeit to 20 cm depth).

It is therefore reasonable to continue to use the Tate et al. (2003) figure as an average value, together with the IPCC default 20 year transition period, for forecasts of the national average change soil carbon stocks following cropland conversion in New Zealand. Nonetheless, it is in fact clear that changes in soil carbon stocks following cropland conversion are actually a strong function of soil and cropland type/practice (see Section 6.2.1.2). Further analysis of the LMI dataset may provide more specifically-applicable stock-change factors.

6.7.2 Stocks and change under current land use/management (1990–2020)

It seems reasonable to argue that carbon stocks in cropland that existed at 1990 are similar to those today for the area of cropland that has remained cropland. Between 1990 and the present date, cropland has increased in area by about 1500 ha per year. This area is likely to lose about 0.5 t C ha⁻¹ yr⁻¹, for about 20 years. On the assumption that cropland conversion continues at the historical rate, primarily because of forage cropping to support intensive pastoral agriculture, it can be expected that equivalent CO₂ emissions related to the losses in carbon stocks through the 2008–2020 period will be about 60 kt CO₂ yr⁻¹, which is only about 1% of New Zealand's present Kyoto commitment to emissions reduction (MfE 2008).

6.7.3 Stocks and change for post-2012 mitigation options

Mitigation options to increase C storage in cropland post 2012 include retention, rather than burning, of cereal residues. However, the most optimistic scenario (that burning can be entirely eliminated and all residues returned to the soil and that the C gain due to residue retention is 0.7 t/ha/year) would result only in increases in soil C storage of about 40 000 t ha⁻¹ annually (see Section 6.6.1). However, during the 2012–2020 period, it is unlikely there will be a large decrease in the proportion of crop residue that is burned and therefore the mitigation potential of this option may not be large. It also needs to be considered that residue retention is likely to result in the need for increased agri-chemical applications, with a resultant increase in fossil-fuel emissions. On the other hand, residue incorporation can reduce over-winter nitrate leaching and may improve air quality.

Adoption of reduced- and no-tillage practices offers several potential benefits to arable farmers, including savings on fuel costs. The merits of no-tillage as a C sequestration option for New Zealand are still open for debate. Until enough reliable local data are available, it may be best to assess the C mitigation potential of no tillage using an average value obtained from the global data base. Based on an increase in soil C of 0.57 t C/ha/year over a 10-year period until a new steady-state C level is attained (West & Post 2002), no-tillage could

arguably increase soil C storage by 5–6 t C/ha. While there may be further expansion in the use of no tillage after 2012, C sequestered in cropland is unlikely to increase by more than 0.5 Mt C even if 50% of land is shifted to no-till.

6.8 Discussion

6.8.1 Implications of forecasts and scenarios

The best current estimates of soil C stocks suggest that carbon stocks in cropland that existed at 1990 are similar to those today for the area of cropland that has remained cropland. Between 1990 and the present date, cropland has increased in area by about 0.5 kha per year. This area is likely to lose about 0.5 t C ha⁻¹ yr⁻¹, for about 20 years, although total losses are only likely to be small (about 1%) in comparison with New Zealand's present Kyoto commitment to emissions reduction.

6.8.2 Effects of information gaps/uncertainties on forecast/scenario reliability

Long-term effects of crop residue management and tillage intensity on soil C storage have not been adequately quantified for New Zealand conditions. Based on available local and international data, improvement in residue management or cultivation practices could, optimistically, increase C storage by perhaps 5–7 t C ha⁻¹. However, the more realistic scenario under New Zealand conditions is that even with residue management, cropland C levels would likely remain approximately static. Without residue management, small losses in soil carbon stocks for areas of cropland converted from pasture are likely to continue over a 20-year period following conversion. Although there is limited information on cropland C for some regions, because the areas involved are small, the resulting uncertainty has relatively little impact on New Zealand's total soil carbon stocks and changes.

6.8.3 Bulk density

One of the most important factors for assessing the impacts of management and land use change on soil C stocks under cropping is soil bulk density. Bulk density data are needed to convert soil C concentrations to soil C stocks (i.e. C mass per volume or area). Owing to the strong influence of soil cultivation on soil structure and density, accurate estimates of bulk density are required to make useful comparisons of soil C stocks under cropping and to assess changes in soil C stocks with changes in land use or management practices. Cultivation also has a strong influence on the placement or distribution of soil organic matter within the soil profile. Consequently, it is important to measure soil C to a depth below the depth of cultivation, to ensure that all actively cycling C is included in the C stock estimate. In soils with deep organic horizons it may be necessary to sample well below the depth of cultivation.

Most comparisons of soil C stocks are based on samples taken to a fixed sample depth. Differences in topsoil bulk density can result from a wide range of management practices including cultivation with different tillage implements to wheel trafficking and treading of livestock on cultivated or pastoral land. Soils with different parent material, textures and mineralogies also tend to have different bulk densities, even under the same long-term land use. Therefore, where there is interest in comparing soil C stocks under different soil types, land uses or management practices with different bulk densities, it is important to ensure the C stock estimates are based on the same mass of soils rather than simply the same sample

depth. Correcting soil C stocks to equivalent masses of soil can result in very different conclusions regarding changes in soil C stocks.

6.9 Conclusions and Recommendations

6.9.1 Present status of studies, datasets, analyses and forecasts

The review of datasets and studies of cropland soil carbon stocks in New Zealand indicates that:

- the Land Management Index (LMI) dataset represents the largest (ca 380 sites) and most comprehensive dataset for quantifying soil C stocks under cropping land across New Zealand (7 regions);
- the LMI dataset was not collected explicitly for the purpose of assessing soil C stocks. A detailed analysis of the LMI dataset should be undertaken to quantify C stocks by soil order and land use intensity in order to improve C stock predictions across New Zealand;
- the LMI dataset also contains detailed information on management history for each site that would allow analysis of the critical management factors impacting on soil C stocks;
- current C stocks in cropland (0–30 cm) were estimated to be 21 million tonnes—effectively identical to the value estimated by Tate et al. (1993) for 1990.

Retaining crop residues rather than burning them, and decreasing the intensity of tillage, are potential management options to retain or possibly increase C stocks in croplands. However, the benefits of neither of these approaches are yet well proven under New Zealand conditions and cannot be strongly endorsed at this time. International work, in particular, has recently raised questions about the presumed effectiveness of non-/low-till systems as a soil-carbon mitigation measure. The MTT dataset could be an important resource to evaluate the benefit of reduced tillage systems for reducing C losses, or sequestering C, under cropping land uses in New Zealand. Funding is urgently needed to continue the trials supporting this dataset.

6.9.2 Key uncertainties, information gaps and research priorities

The major limitation of New Zealand cropping datasets for C accounting purposes is the general absence of data for the 15–30-cm depth. Analysis of some of the newer datasets, including the LMI dataset, will enable us to fill this knowledge gap. Reduction in tillage intensity and/or retention of crop residues may be the best management options to increase C in cropland, but further work is needed to quantify the C mitigation potential of these practices under New Zealand conditions. At present, there is no on-going research planned in this area. The Millennium Tillage Trial is expected to provide some key initial information to quantify the long-term effects of reduced- and no-tillage management under New Zealand conditions. Given the importance of no-/low-till practices as potential mitigation practices, expanding this study to a wider range of soil/climate regimes is a high priority.

In addition it is noted that:

- while many of the major cropping soils are well represented in the datasets that are available for further analysis, some important gaps remain for intensive cropping soils, particularly in the Gisborne, Manawatu, Wanganui and possibly Waikato regions;
- measurements are required of soil C stocks well below the depth of cultivation to ensure that continuous long-term cropping is not impacting on soil C stores deeper in the soil profile;
- expansion of intensive cropping onto pastoral lands is expected to result in increased losses of soil C, for example, associated with such activities as expansion of intensive vegetable cropping in the Waikato. Analysis of the LMI dataset should be extended to investigate likely impacts of these activities;
- it is not well-known how the increased use of irrigation on former dryland farming systems is impacting on soil C stocks during conversion from pasture to cropping, although the current evidence is that losses could be expected;
- the increased use of forage cropping for supplementary feed production, and the increased frequency of pasture renewal (especially where cultivation is involved) in intensive pastoral farming systems (dairying), has the potential to reduce soil C stocks significantly. At present it remains unclear as to the area over which this is occurring. Analysis of the LMI dataset should be extended to investigate the likely impacts (per unit area) of these activities;
- the contribution of plant roots to soil C is still poorly understood, though they represent the major C input where above-ground residues are burned or removed. The possibility of increasing soil C by developing crop plants with larger root mass may be worth considering.

6.9.3 Implications of accounting and mitigation options for New Zealand's post-2012 net position

On the basis of present evidence, it seems reasonable to assume that for those land areas that have been cropland since 1990, current carbon stocks are similar to those in 1990. Since 1990, the cropland area has increased by about 500 ha per year. This area is likely to lose about $0.5 \text{ t C ha}^{-1} \text{ yr}^{-1}$, for about 20 years. On the assumption that cropland conversion continues at the historical rate, primarily because of forage cropping to support intensive pastoral agriculture, annual losses in carbon stocks through the 2008–2020 period can be expected to be about $0.02 \text{ Mt CO}_2 \text{ yr}^{-1}$.

This small loss could potentially be offset by adopting retention of harvest residues and no/low-till cultivation practices as mitigation practices. However, at present it is difficult to recommend such practices on the basis of emissions reduction alone, not just because the total mitigation potential is small, but also because:

- gains in soil carbon from residue incorporation may not be large under New Zealand conditions;

- elimination of harvest residue burning may increase the need for increased agricultural applications, with associated increasing fossil fuel emissions that may offset gains from great C stocks additional emissions from burning;
- no-/low-tillage may increase carbon stocks in some but not in all circumstances, and in New Zealand's moister climate overall efficacy remains unproven.

On the other hand, residue incorporation can reduce over-winter nitrate leaching and improve air quality, and adoption of no-tillage can decrease fossil-fuel consumption and associated greenhouse gas emissions, and increase water-use efficiency, thus allowing greater water conservation, and reduce soil erosion (and loss of carbon in eroded soil) on erosion-prone land. By considering the total effects of resource sustainability, soil quality and emissions reductions considerations, these associated benefits strengthen the justification for residue retention and no-/low-till cultivation.

7. Effects of Horticulture on Soil Carbon

Markus Deurer (HortResearch), Brent Clothier (HortResearch)

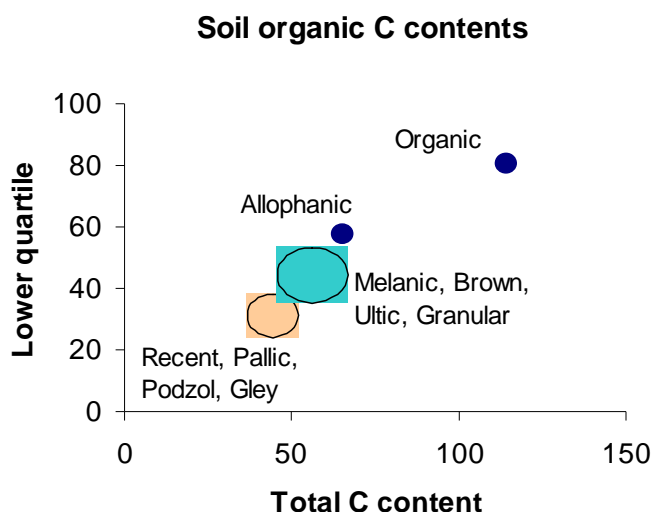
7.1 Introduction

7.1.1 The role of soil carbon management in soils under orchard/vineyard land use

Soil carbon stocks are determined by soil type, climate, type of land use and management. For soil carbon classification purposes, it is useful to distinguish among the impact of inherent soil properties, soil genoform, and anthropogenic influences—the soil phenoform (Droogers & Bouma 1997).

A recent document (Mackay et al. 2006) gives an overview of the typical range of soil carbon stocks associated with New Zealand's most prominent soil genoforms, the soil orders. The different New Zealand soil orders have quite different soil carbon stocks (Fig. 7). Soil carbon stocks in 0–0.15 m depth vary by about a factor of four in the different soil orders. Therefore, independent of management or land-use history, orchards/vineyards located on different soil series will have different soil carbon stocks. To our knowledge, there has been no systematic survey of the land areas of orchards/vineyards associated with specific soil orders in New Zealand.

Figure 7 Mean total and lower quartile carbon (C) contents (t/ha) of soils in the 500 Soils Project data set (n=700; Mackay et al. 2006) in the top 0.15 m depth. Not all soil orders are represented.



Unfortunately, to our knowledge, there has been no systematic survey of the land areas of orchards/vineyards associated with specific soil orders.

We term the anthropogenic influence on soil carbon stocks as “soil carbon management”. Soil carbon management includes all “land management practices that maintain or increase soil C” (Kimble et al. 2007). This definition is very similar to the definition of “carbon sequestration management”, namely “... any management practice that increases the photosynthetic input of carbon and/or slows the return of stored carbon to CO₂ via respiration, fire, or erosion will increase carbon reserves, thereby sequestering carbon or building carbon sinks” (Smith et al. 2007).

The various options for soil carbon management and their potential to increase soil carbon stocks under arable and forest land-use have been reviewed by Lal (2004) and Smith et al. (2007). However, to our knowledge, no such review has been carried out on options for soil carbon management in orchards and vineyards.

With respect to soil carbon management in orchard/vineyards, we suggest distinguishing between traditional/integrated systems and organic practices.

7.1.2 Soil carbon management in integrated orchards/vineyards

Under traditional/integrated systems, no deliberate practices for soil carbon management exist. Soil carbon is not yet valued by such orchardists/winegrowers as providing additional value to their “natural capital” with respect to production and economic outcomes. Quite the opposite in fact, because in many vineyards, the nutrient source of mineralised nitrogen (N) from soils with high soil carbon stocks could lead to increased vegetative vigour, which could reduce grape yield and quality, and most certainly require more labour-intensive practices such as pruning and hedging. Therefore, winegrowers prefer to use synthetic fertilizers, as their availability for the vines is much easier to predict and manage than the nutrient supply from organic matter sources such as composts, or manures.

Another example of a ‘negative’ soil carbon management practice is the avoidance of cover crops for the entire orchard/vineyard floor because cover crops compete with the economic crops for water and nutrients (Tworkoski & Glenn 2001; Tesic et al. 2007). Economic incentives such as carbon credits (Sparling et al. 2006), or market access regulations that reward environmental stewardship, such as the GlobalGAP protocols, might in the future change the lack of consideration of ‘positive’ soil carbon management in traditional/integrated orchards/vineyards.

This is fundamentally different from the situation under arable land use, where a positive effect of soil carbon on productivity has been reported for New Zealand by the Sustainable Land Use Research Initiative (Mackay et al. 2006).

7.1.3 Soil carbon management under organic orchards/vineyards

Soil carbon management is the key for success in organic orchards/vineyards. In organic orchards/vineyards, soil organic matter is the main source of the major plant nutrients, and for nitrogen, it is the only means of supply. Because of the soil-ecosystem service of nutrient supply, organic orchardists/winegrowers value soil organic matter highly. Better soil carbon management in organic orchards/vineyards than in traditional/integrated ones should, therefore, provide a ‘positive’ outcome. Indeed, for arable systems it was found that carbon inputs to the soil are generally higher in organic than in integrated or conventional production systems (Gunapala & Scow 1998; Fliessbach et al. 2007).

However, the sequestration objective of soil carbon management somewhat contradicts the nutrition objective, using controlled and continuous soil carbon turnover as a nutrient source in organic orchard/vineyard systems. This dilemma was recently discussed in the scientific literature under the title: The soil carbon dilemma – should we hoard it or should we use it? (Janzen 2006).

7.2 Review of New Zealand Datasets and Studies

7.2.1 Introduction

To the best of our knowledge, there are no large datasets available for horticultural land use, either in New Zealand or internationally, that are explicitly dedicated to the monitoring of soil carbon stocks, and their change over time due to change in land use, and/or management practices.

We have identified two New Zealand datasets (Datasets 1 and 2, below) that can be used to provide a preliminary estimate for the dynamics of soil carbon stocks under horticultural land use. Two further datasets (Datasets 3 and 4) could be extended in the future to serve that purpose as well. All datasets are held by HortResearch and comprise the following:

- Dataset 1:** Soil carbon management of apple orchard systems (organic/integrated) in Hawke's Bay
- Dataset 2:** Soil carbon status and management of vineyard systems in Marlborough
- Dataset 3:** Soil carbon status of vineyards in Hawke's Bay
- Dataset 4:** Soil carbon status of kiwifruit orchards in the Bay of Plenty, Nelson, and Northland.

The datasets are described in detail in the Appendix to this Chapter. Here, we focus only on the four key characteristics of the datasets that are relevant for the analysis of the soil carbon stocks, and their change over time under orchard/vineyard practices. These characteristics are:

- (i) *How representative is the dataset?*
What is the total number of soil samples, the number of locations/sites, the soil order and/or the number of soil orders sampled?
- (ii) *Is it possible to derive soil carbon stocks from the dataset?*
What is the sampling depth(s), and have bulk densities been measured?
- (iii) *Is it possible to derive the change of soil carbon stocks over time?*
Is there information about the land use before conversion to horticulture and its associated soil carbon stocks. Is there information about the number of times or dates when soil carbon stocks were sampled?

- (iv) *What impact has a specific soil carbon management had on the rate of change of soil carbon stocks?*

Is there detailed information on the soil carbon stocks (see item 2 above), their change over time (see item 3 above), and the soil carbon management over at least the last 10 years?

Additionally, we give a short description of the primary objective of the study that resulted in the dataset, and we list the references of any publications where results can be found.

We also summarise the only known published New Zealand study on soil carbon change under horticulture: a comparison of carbon stocks in an organically versus non-organically managed apple orchard.

7.2.2 Review of New Zealand datasets

Dataset 1: Soil carbon management of apple orchard systems

The primary objective of the work that led to this dataset was to investigate the impact of soil carbon management on soil quality. The major results have recently been published by Deurer et al. (2008a).

How representative is the dataset?

Six soil samples were taken in 2006 from each of the tree row and the alley between the tree rows in an organic and an adjacent integrated apple orchard in Hawke's Bay.

Both sites have soils that belong to the same soil order (Recent). Therefore, because of the small number of samples and the single soil order sampled, the dataset is not necessarily representative of the soil carbon stocks of apple orchards in other parts of Hawke's Bay, or New Zealand. However, given the need for well-drained soils for apple production, Recent soils probably represented a large proportion of orchard soils.

Is it possible to derive soil carbon stocks?

Yes, soil carbon stocks and bulk densities were measured in three incremental depths: 0–0.1, 0.1–0.2, 0.2–0.3 m.

Is it possible to derive the change of soil carbon stocks over time?

Yes, but in a limited way. The land use before horticulture was (arable) commercial vegetable production in both systems, but the soil carbon stocks at the point of land-use change were not measured. However, when the apple trees were planted about 12 years ago, the alleys between the apple trees were sown into pasture. We assume that 12 years is long enough for soil carbon in the top 0.3 m of the pasture system to reach a new equilibrium. Therefore, the difference of soil carbon between the tree rows and the alleys can be used to estimate the change of soil carbon stocks due to the change from pasture to apple orchard.

What impact has/had a specific soil carbon management on the rate of change of soil carbon stocks?

The information on soil carbon stocks, their change over time, and the soil carbon management practices over the last 12 years is known.

Dataset 2: Soil carbon status and management of vineyard systems in Marlborough

The primary objective of the work leading to the compilation of this dataset was to investigate the effect of different management practices on soil carbon stocks in the vine rows. The major results of this work were presented at a wine workshop.

How representative is the dataset?

Three soil samples were taken in 2007 from each of 2–4 sites in the vine-row and inter-row of five different vineyards in Marlborough. The sites were all of the same soil order (Recent). Therefore, because of the small number of samples and the single soil order sampled, the dataset is not necessarily representative of the soil carbon stocks of vineyards in the whole of the Marlborough region, or of New Zealand. Vineyards, because of their requirement for free-draining conditions do, however, tend to be on Recent soils.

Is it possible to derive the soil carbon stocks?

Yes, but only for 0–0.15 m and only for one vineyard. Soil carbon stocks and bulk densities were measured in a mixed sample from 0–0.15 m depth.

Is it possible to derive the change of soil carbon stocks over time?

Yes, but in a limited way. The land use before horticulture was extensive sheep and beef pasture, but the soil carbon stocks at that point in time had not been measured. However, the inter-rows and headlands in the vineyard continued to be used as pasture. Therefore, the difference in soil carbon between the tree rows and the inter-rows/headlands can be used to estimate the change of soil carbon stocks from extensive sheep and beef pasture to vineyard.

What impact has a specific soil carbon management had on the rate of change of soil carbon stocks?

The information on soil carbon stocks, their change over time, and the soil carbon management over the last 10–15 years is known.

Dataset 3: Soil carbon status of vineyards in Hawke's Bay

The primary objective of the work leading to the compilation of this dataset was to begin to investigate the components of terroir. The soil carbon status was included as part of this study in Hawke's Bay. The dataset has not been presented or published elsewhere.

How representative is the dataset?

Three soil samples were taken in 2007 from the vine-row of five different vineyards. Neither the soil order nor soil type of the sites is known. Therefore, because of the small number of samples the dataset is not necessarily representative of the soil carbon stocks of vineyards in either Hawke's Bay, or in New Zealand.

Is it possible to derive the soil carbon stocks?

Yes, but only for 0–0.2 m. The soil carbon stocks and bulk densities were measured in a mixed sample from 0–0.2 m depth.

Is it possible to derive the change of soil carbon stocks over time?

No.

What impact has a specific soil carbon management had on the rate of change of soil carbon stocks?

The dataset contains too little information to answer this question.

Dataset 4: Soil carbon status of kiwifruit orchards in the Bay of Plenty, Nelson, and Northland

The primary objective of the work leading to the compilation of this dataset was to analyse the soil carbon status in different kiwifruit orchards as part of a study on vine nutrition. The dataset has not been presented or published elsewhere.

How representative is the dataset?

Three soil samples were taken in 2006 and 2007 from the vine row of each of five different kiwifruit orchards. Four of these orchards were in the Bay of Plenty, one was in Nelson, and one in Northland. The soil order of the sites is not known. Therefore, because of the small number of samples the dataset is not necessarily representative of the soil carbon stocks of kiwifruit orchards in New Zealand.

Is it possible to derive the soil carbon stocks?

Only for the samples from the Bay of Plenty, and only for 0–0.2 m. For the other samples, no bulk densities were measured.

Is it possible to derive the change of soil carbon stocks over time?

No.

What impact has a specific soil carbon management had on the rate of change of soil carbon stocks?

The dataset contains too little information to answer this question.

7.2.3 Summary of New Zealand studies

Deurer et al. (2008a) have published the only study that compared soil carbon levels under different management regimes in New Zealand orchards. This study compared soil carbon under apple orchards in response to organic and non-organic treatments and drew on the data summarised in Dataset 1. It compared soil carbon stocks under integrated (i.e. normal, non-organic) and organic apple orchards that have the same soil order, type, texture, and climate; and have also had the same orchard management for at least 10 years). Soil carbon stocks under the grassed alleys in the orchards were used as a “permanent pasture” reference. The study found that soil carbon to 30 cm depth declined under both organic and non-organic management systems, but with smaller losses under the organic system—by about 5 t C ha⁻¹. The tree rows of the integrated orchard were herbicided, drip-irrigated, and received no external organic matter inputs. The tree rows of the organic orchard were grassed, not irrigated, and regularly received compost addition.

7.3 Review of Key International Information

Very few international studies have examined soil carbon stocks and change under horticulture.

In the only study for which any detail is available, four pairs of conventional/organic vineyard soils (0–0.1 m) in Germany and France on different soil types were compared (Probst et al. 2008). No significant differences of soil organic carbon stocks were found. However, the vine-rows in this organic system were not covered by grass or other crops:

organic management (≥ 10 years) meant only excluding the use of synthetic pesticides, using a combination of organic fertilizers, green manure, instead, and using shallow or reduced tillage.

Montanaro et al. (2008) reported a study (available only as a conference abstract) in which they measured the net gain of carbon (entire system: above and below ground) in peach and kiwifruit orchards. They found that sustainable soil management (namely, cover crop, no-tillage, compost application, mulching of pruning residues, regulated deficit irrigation) led to a net carbon gain of 17 t/ha/year, while the comparable conventionally managed orchards lost 6 t/ha/year.

McGourthy and Reganold (2004) reported (available only as a conference abstract) that they found cover crops in the inter-rows of vineyards increased soil organic carbon stocks. However, no quantitative data were provided.

7.4 Mitigation Opportunities for Horticultural Soils

7.4.1 Assessment of mitigation options

Cover crops

The introduction and/or efficient management of cover crops in the alleys and tree rows of orchards seems to be the most promising and cost-effective option for soil carbon management in orchard/vineyard systems. We consider the use of permanent grass as the cover crop in the tree row is the major reason why the organic apple orchard in Hawke's Bay had significantly higher soil carbon stocks than the comparable integrated system (Deurer et al. 2008a).

Vineyard floor management is already an integral component of integrated production in Europe, North America, South Africa, and New Zealand (Testic et al. 2007). As well as its potential for increasing carbon stocks, orchardists and grape-growers can use it as a powerful tool to control unwanted vegetative growth. However, under hot and dry conditions, and especially without additional irrigation, competition for water and nutrients (particularly at sensitive stages such as bloom and berry set for grapes) under such orchard-floor management can lead to a substantial decrease in yield (Testic et al. 2007).

A further step might be to introduce purpose-bred grass species with higher productivity, and carbon allocation to deeper roots. For example, establishing deep-rooted grasses in savannahs has been reported to produce very high rates of soil carbon accrual (Fisher et al. 1994). The grass species could be complemented by nitrogen-fixing plants such as legumes. Introducing legumes into grazing lands can promote soil carbon storage through enhanced productivity from the associated N inputs (Soussana et al. 2004).

Organic orchard/vineyard systems

Management of an orchard/vineyard as an organic system involves the use of cover crops, along with other elements of soil carbon management such as the application of compost, mulches and/or manures. In the study comparing one organic and one integrated apple orchard in Hawke's Bay, soil carbon stocks were significantly higher in the organic than in the integrated system (Deurer et al. 2008a). However, we are not aware of any systematic studies comparing a larger number of organic and integrated orchard/vineyard systems, which could confirm that organic systems have better soil carbon management.

Many studies have shown an increase in soil organic carbon stocks under organically managed arable farming. It is highly likely that a similar result would be observed for organically-managed orchard/vineyards.

A regional soil survey in the Netherlands showed that land use across a period of 63 years had a distinct effect on soil organic carbon stocks within one specific soil series. Soil organic carbon stocks significantly increased in soils under permanent pasture and arable crops that were organically managed (Pulleman et al. 2000). Carbon inputs into the soil were generally higher in organic than in integrated or conventional production systems (Gunapala & Scow 1998; Fliessbach et al. 2007).

However, the seemingly better carbon sequestration under “organic” management might have hidden carbon costs. For example, no net sink for carbon is likely to accompany the use of animal manure/compost on agricultural lands, as this in most cases requires ‘mining’ carbon somewhere else (Schlesinger 2000).

Carbon sequestration in subsoil horizons

Routine soil carbon stock inventories estimate the soil organic carbon pool down to a soil depth of about 1 m. It has been speculated that deeper soil horizons may have a high capacity to sequester soil organic carbon, as the turnover time and chemical recalcitrance of soil organic matter increases with depth. Subsoil carbon sequestration may be achieved by higher inputs of fairly stable organic matter to deeper soil horizons. Lorenz and Lal (2005) consider that the subsoil might have considerable capacity for carbon storage and suggest that using breeding and genetic engineering will result in better efficiencies of carbon placement at depth. This can be achieved directly by selecting plants and cultivars with deeper and thicker root systems that are high in chemically recalcitrant compounds like suberin. Furthermore, increasing the amount of recalcitrant compounds could be a target for plant breeding and biotechnology to promote soil carbon sequestration.

Another way to achieve subsoil carbon sequestration is to promote a high surface input of organic matter. That would promote the production of dissolved organic carbon DOC that can be transported to deeper soil horizons. However, this is only likely to result in permanent increases if drainage at depth is limited.

7.4.2 Biochar as an emissions offset option

Introduction

Biochar is a type of charcoal produced from biomass. The use of biochar as a soil carbon sequestration strategy relies on charcoal produced as a residue of pyrolysis of biomass. Under complete or partial exclusion of oxygen, ‘waste’ biomass is heated to moderate temperatures, usually between 400 and 500°C (namely low temperature pyrolysis), yielding fuel energy, and biochar as a carbon-rich and more stable by-product.

Biochar seems especially well suited for use in orchard/vineyard systems. For example, the addition of biochar should not increase vegetative vigour. As opposed to other biomass-derived carbon materials (e.g., compost), biochar is not easily decomposed. Consequently, the application of biochar does not increase the availability of nutrients, such as nitrogen. Biochar could also improve the efficiency of fertilizers use, and possibly reduce the leaching

of nitrogen and phosphorus, thus improving the overall eco-efficiency of nutrient management in orchards/vineyards (see below).

The use of biochar as a potential strategy for soil-carbon sequestration has recently been discussed nationally, and internationally. Biochar needs to fulfil at least four criteria to be a successful strategy for soil carbon sequestration in orchards/vineyard systems:

- (i) The half-life of biochar that is incorporated into soil needs to be at least 100 years. This is the criterion for any strategy to be considered as a viable soil carbon sequestration approach under regulatory frameworks such as the proposed PAS 2050.
- (ii) The use of biochar results in a net reduction of equivalent CO₂ emissions for a horticultural enterprise. A full life cycle analysis, including the energy needed for its production, transport and incorporation into the soil thus needs to be considered.
- (iii) Biochar needs to become locally available at a cost-effective price.
- (iv) It must be possible to incorporate large amounts of biochar into soils without compromising the product yield and quality in orchards/vineyards in either the short or the long term, or lead to other environmental problems.

Stability of biochar in soils

Large accumulations of charred material with residence times in excess of 1000 years have been found in soil profiles (Saldarriaga & West 1986; Glaser et al. 2001; Forbes et al. 2006). Several authors (e.g., Glaser et al. 2003) have reported large stocks of pyrogenic black carbon, such as the Amazonian dark earths or terra preta, several hundred years after the cessation of the activities that added it to the soil. This is due to its chemical recalcitrance.

However, very little is known about the half-life of specific types of 'industrial' biochar. The recalcitrance of biochar in soils depends on a multitude of factors, including the type of biomass used for pyrolysis, the pyrolysis conditions, soil properties, and local climate. Typically, however, the half-life of biochar from low-temperature pyrolysis is longer than 100 years (Lehmann et al. 2006; Singh & Cowie 2008).

It is possible that soil micro-organisms may adapt to the use of biochar as a carbon source, which might shorten its half-life. For example, there are micro-organisms that use biochar as their sole carbon source (Hamer et al. 2004).

Net reduction of equivalent CO₂ emissions due to the use of biochar

Compared with other biomass-derived carbon (e.g., compost), biochar leads to a reduction of equivalent CO₂ emissions from soils due to its long half-life. However, the slow pyrolysis-based bioenergy systems produce not only biochar for soil carbon sequestration, but also energy. Gaunt and Lehmann (2008) showed that the use of biomass for energy production and soil carbon sequestration can have a combined benefit of about 2–19 t CO₂e ha⁻¹ year⁻¹. Of these avoided emissions, 41–64% are related to the retention of carbon in biochar, with the rest to offsetting fossil fuel use for energy, fertilizer savings, and avoidance of soil emissions other than CO₂, such as nitrous oxide (Gaunt & Lehmann 2008).

The proportion of carbon retained in biochar during pyrolysis varies with pyrolysis temperature and the type of biomass (Lehmann et al. 2006). A typical level of carbon

recovery is 50% of the initial carbon content. This carbon has a typical half-life of more than 100 years (Lehmann et al. 2003, 2006).

Practicality and cost-effectiveness of biochar use

Currently, there is no large-scale facility for low-temperature pyrolysis in New Zealand, although this might change in the future. Another question with respect to the practicality of the use of biochar relates to how much biochar can be effectively and practically applied to soils.

From the data available for highly weathered tropical soils, it appears that crops respond positively to biochar additions of up to at least 50 t C ha⁻¹ (Lehmann et al. 2006). For most plant species and soil conditions, this maximum was not reached even with 140 t C ha⁻¹, and growth reductions may only occur at even higher application rates (Lehmann et al. 2006). We note, however, that most knowledge is derived from experiments with highly weathered tropical soils that have very low natural soil organic carbon contents. Little is known about the effect of biochar additions on relatively more fertile soils in a temperate climate.

The cost of incorporating biochar in soil, instead of using biomass solely for electricity generation, was estimated as \$US47 per t CO₂ contained in biochar (Gaunt & Lehmann 2008). However, this did not incorporate the additional costs associated with the transport of biochar from the pyrolysis plant to the site of application, and its incorporation into the soil. Currently, the market price for CO₂ is \$US9–16 per t CO₂, so incorporation of biochar into soils is not yet a cost-effective option (Gaunt & Lehmann 2008). However, the carbon prices and emissions-trading costs could be much higher in the future. For example, in the European Union Emission Trading Scheme the price is \$US20, and would lie around \$US25–85 if the social costs of climate change were used as the basis for calculations (Stern 2007).

Possible short- or long-term consequences of biochar applications

No published data are available on any possible negative consequences of biochar applications to soils, at least not from those based on field-scale studies, but field-scale studies of biochar incorporated into soils have only recently started. Below, we provide an assessment of the potential risks that have not yet been thoroughly evaluated, especially under the conditions of New Zealand's soils and climate.

The type of biomass and pyrolysis conditions can modify the amount and composition of phytotoxic and potentially carcinogenic organic materials that are a by-product of pyrolysis (Lima et al. 2005).

Biochar also contains aromatic and aliphatic organic compounds that may cause, or enhance, soil water repellency. Many New Zealand soils have been found to be water repellent after dry summers which causes reduced uptake of rain fall and a consequent decrease in pasture growth (Deurer et al. 2008b). Water and nutrients may instead flow into surface waters as another deleterious consequence of water repellency (Doerr et al. 2000). Many topsoils in New Zealand already have very high carbon contents, and carbon content is generally positively correlated with the occurrence of water repellency (Doerr et al. 2000). No studies have yet been undertaken to investigate if biochar could cause, or enhance, soil water repellency. However, water repellency was reported to occur in reclaimed mine soils that contain sandy sediment mixtures with significant proportions of lignite (brown coal) (Doerr et al. 2000). Another indication of the potential risk of using biochar and causing soil water

repellency is that hydrophobicity often occurs in topsoils after forest fires, which in a way ‘mimics’ pyrolysis (Doerr et al. 2000).

Biochar might also interact with existing soil carbon in unexpected ways. For instance, Wardle et al. (2008) prepared charcoal and mixed it with the forest soils in contrasting forest stands in northern Sweden for 10 years. Micro-organisms significantly increased in the biochar-treated soil. As a consequence, some of the original soil organic matter was lost and net soil carbon sequestration was small. It is not known how this finding might relate to productive enterprises, such as orchards/vineyards.

Impact of biochar on physical and chemical soil properties

It has been found that in highly-weathered, coarse-textured soils, biochar improves the soil’s filtering and buffering capacity for nutrients. Biochar adsorbs more cations per unit carbon than most other types of soil organic matter, due to its greater surface area, greater negative surface charge, and greater charge density (Liang et al. 2006). However, the magnitude of the cation-exchange capacity depends on the type of biomass from which biochar is produced, and the pyrolysis conditions. Biochar properties can also change considerably with time during exposure to the soil environment (Lehmann 2007).

Biochar retains nutrients, especially nitrogen and phosphorus (Glaser et al. 2002; Lehmann et al. 2003), and increases the nitrogen fertiliser-use efficiency for plants (Chan et al. 2007). Biochar was found to reduce the leaching of nitrate, ammonium, phosphorus and other ionic compounds (Beaton et al. 1960; Radovic et al. 2001; Lehmann et al. 2003; Mizuata et al. 2004). It has been observed that biochar absorbs hydrophobic organic contaminants (Gustaffson et al. 1997; Accardi-Dey & Gschwend 2002).

In highly weathered, coarse-textured soils, biochar improves the soil’s water retention capacity. In Amazonian charcoal-rich anthrosols, the field water-retention capacity was 18% higher than for surrounding soil without charcoal (Glaser et al. 2002). However, in another study (Tryon 1948) with three different soil textures (sandy, clayey and loamy), charcoal was found to increase plant-available water contents only in the sandy soil, but had no effect in the loamy soil, and decreased it in the clayey soil.

Implication of biochar sequestration for non-CO₂ emissions and removals

In greenhouse experiments, with biochar additions of 20 g per kilogram of crop, N₂O emissions were reduced by 50% in a soybean crop, 80% in a forage grass stand; methane emissions were completely suppressed in both (Rondon et al. 2005). A reduction of N₂O emissions was also found in short-term incubation experiments (Yanai et al. 2007). The reduction of N₂O emissions may be a consequence of better aeration of the soil, along with a shift in the C:N ratio.

7.5 Current Best Estimates of Horticultural Soil Carbon Stocks and Change

7.5.1 Carbon stocks and change

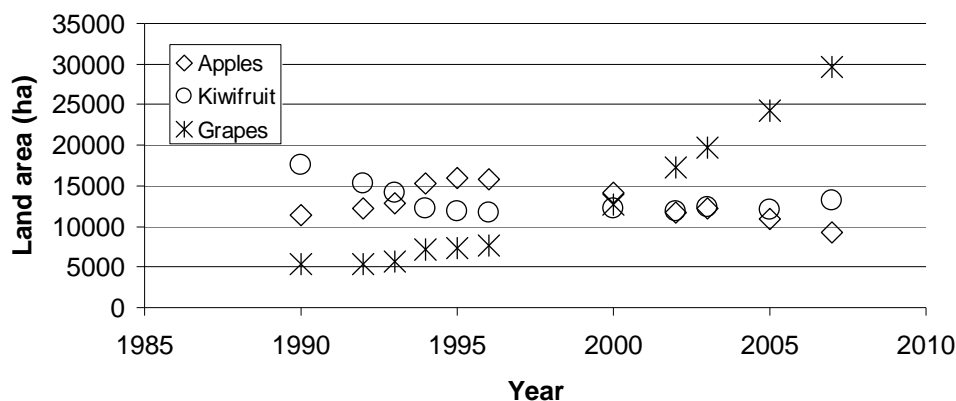
Activity data

National area of land use conversion from permanent pasture to orchards/vineyards

Tate et al. (2005) estimated that around 3000 ha were being converted annually from permanent pasture to orchard/vineyards. However, the total area of all “outdoor fruit crops” was 46 808 ha in 1990 and 67 000 ha in 2007 (source: Statistics New Zealand), for an

average increase of about 1200 ha per year. There have also been recent more dynamic developments in the horticultural sector. In particular the area of wine grapes has increased quite sharply at a linear rate of about 2400 ha per year since 2000 (Fig. 8). Over this time there were relatively small changes in the area of the two major horticultural crops, apples and kiwifruit.

Figure 8 Land areas in New Zealand associated with the three largest horticultural crops (kiwifruit, apple, and wine grapes).



National area associated with specific soil carbon management in orchards/vineyards

Around 1500 ha of orchards/vineyards are currently classed as organic. This is about 2.5% of the total area. However, the organic sector is also highly dynamic and has increased by about 160% over the last decade.

No estimates are available on the areas of orchards/vineyards associated with specific soil carbon management practices such as cover crops, application of compost, and mulching.

Stocks and change under current land use/management (1990–2020)

Change of soil carbon stocks as a result of conversion of pasture to horticulture based on available data¹⁹

First, we estimated the change in soil carbon stocks in two comparable apple orchards in Hawke's Bay (integrated/organic) with respect to the permanent pasture reference (Dataset 1). Using the soil under the permanent grass in the alley of each orchard as a reference, we considered the soil carbon stocks in 0–0.3 m depth. More details on the soils and orchard management are given elsewhere (Deurer et al. 2008a).

Over the 12 years from 1994 to 2006, the row in the organic apple orchard system lost about $1.7 \pm 3.1 \text{ kg C/m}^2$ (Table 1). Assuming that (at most) half the total area of the orchard is managed as a row and the rest is permanent pasture, this equals a decline in the carbon stocks by $8.5 \pm 15 \text{ t C ha}^{-1}$. This value corresponds closely to the estimated loss of $9 \pm 7 \text{ t C ha}^{-1}$ of

¹⁹ In this and all following sections on soil carbon change under horticulture, the number after a “±” sign denotes one standard deviation, rather than one standard error. However, for the $-9 \pm 7 \text{ t C ha}^{-1}$ of Tate et al. (2005), the number after the “±” sign denotes one standard error.

Tate et al. (2005). We also note that because of the high standard deviation of the measurements, the change in soil carbon could also be zero or positive.

Over the same time, the row in the integrated apple orchard system lost about $22 \pm 13 \text{ t C ha}^{-1}$ (Table 20). Assuming again that only half the total area of the orchard is managed as a row and the rest remains as permanent pasture, this equals a decline in the carbon stocks by $11 \pm 7 \text{ t C ha}^{-1}$. The average value is somewhat higher than the estimated loss of $9 \pm 7 \text{ t C ha}^{-1}$ of Tate et al. (2005), but not in any statistically significant way.

Expressed differently, soil carbon stocks in 0–0.3 m depth were higher in the organic apple orchard in Hawke’s Bay, at $86.5 \pm 16 \text{ t C ha}^{-1}$, than the $70 \pm 6.5 \text{ t C ha}^{-1}$ in the comparable integrated apple orchard. However, the differences were statistically significant only at 0–0.1 m and the 0.2–0.3 m depths, but not for the full the 0–0.3 m depth. It is not clear how much of this difference between the organic and integrated system can be attributed to the soil management option of “cover crop” or “organic” practices.

In orchards with no grass or cover crops in the alley, the soil carbon stock losses have to be multiplied by 2.

Table 20 Average carbon (C) stocks and their estimated change in two apple orchard soils (organic, integrated) in Hawke’s Bay (Dataset 1, Section 7.2.2). The alley in both systems is permanently covered with grass and served as the reference

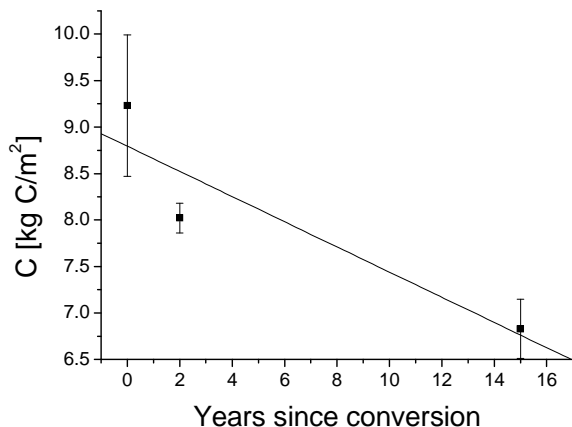
Depth (m)	Organic – row (kg C m ⁻²)	Organic – alley (kg C m ⁻²)	Integrated – row (kg C m ⁻²)	Integrated – alley (kg C m ⁻²)
0-0.3	7.8±1.9	9.5±1.2	5.9±0.6	8.1±0.7
	Estimated change in soil organic stocks over 12 years (kg C m⁻² yr⁻¹)			
	-0.14±0.3	0	-0.18±0.1	0

Second, we estimated the change in soil carbon stocks in one vineyard in Marlborough with respect to a permanent pasture reference (Dataset 2). We used the soil under the permanent grass in the headland of the vine-rows as this reference.

Over the last 15 years (1991–2006) the row in the integrated vineyard system lost about $2.4 \pm 1 \text{ kg C m}^{-2}$ in 0–0.15 m depth (Fig. 9). Assuming that (at most) half the total area of the vineyard is managed as a row and the rest is permanent pasture, this equals a decline in the carbon stocks by $12 \pm 5 \text{ t C ha}^{-1}$ in 0–0.15 m depth. This value is somewhat higher than the estimated loss of $9 \pm 7 \text{ t C ha}^{-1}$ of Tate et al. (2005), though the difference is not statistically significant.

In vineyards with no grass or cover crops in the alley, the soil carbon stock losses reported above have to be multiplied by 2. It is not clear how much soil carbon was additionally lost in 0.15–0.3 m depth. Unfortunately, there are no data yet on the change of soil carbon stocks with time under different management systems (e.g., organic) in vineyards.

Figure 9 Average carbon stocks and their estimated change in one integrated vineyard in Marlborough (from Dataset 2). The average rate of change of the soil carbon stocks (0–0.15 m depth) is $-0.14 \pm 0.06 \text{ kg C m}^{-2} \text{ yr}^{-1}$. The alley and the headland in the vineyard were permanently covered with grass and served as the reference (“Year 0”).



Change of soil carbon stocks as a result of mulching of soil surface under vine-rows based on available data

In one integrated vineyard in Marlborough for the 0–0.15 m depth, soil carbon stocks were $49.5 \pm 3.5 \text{ t C ha}^{-1}$ without mulching, and $63.5 \pm 13.5 \text{ t C ha}^{-1}$ with mulching (from Dataset 2, Section 7.2.2). Therefore, the use of mulch in this one vineyard led, on average, to higher soil carbon stocks, but this difference was not statistically significant.

In the organic vineyard in Marlborough for the 0–0.15 m depth, the headland (i.e. the permanent pasture reference) had $7.6 \pm 0.9 \text{ kg C m}^{-2}$ and the respective vine-row had $8.1 \pm 0.6 \text{ kg C m}^{-2}$ (Dataset 2, Section 7.2.2). Therefore, in this one vineyard the soil carbon stocks under an organic management system were not significantly different from those in the permanent pasture reference. That is, there was no significant loss of soil carbon under this organic vineyard system.

In our opinion, all estimates given in this section are very uncertain.

Change of soil carbon stocks as a result of conversion of pasture to horticulture based on modelling

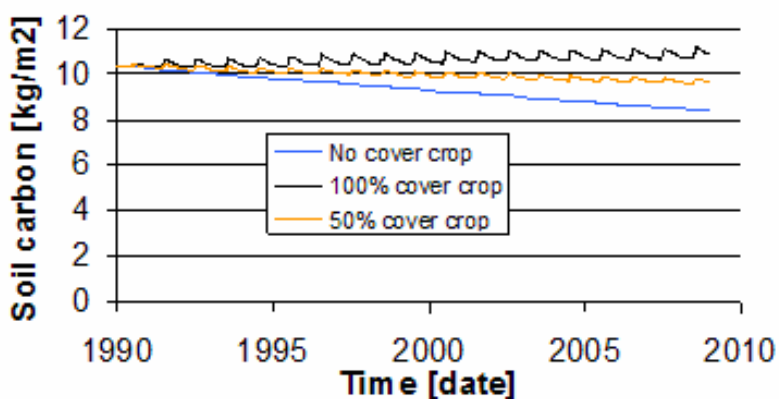
A recent study (Tate et al. 2005) estimated the land-use effect of a conversion of grazing land to horticulture on soil organic carbon stocks for the 0–0.3 m depth. They estimated a total change of $-9 \pm 7 \text{ t C ha}^{-1}$. The estimate of the land-use effect came from a model with no land-use–soil–climate interactions, and, therefore, they applied it across all soil-climate categories and slope-rainfall combinations (Tate et al. 2005). In New Zealand, there are some measurements of soil carbon changes with conversion to and from pastures for some grain crops, but for horticultural conversions, there are only the studies listed above and none of those had been available to Tate et al. (2005). The uncertainty of these estimates is noted to be high.

We also used the HortResearch SPASMO (Soil Plant Atmosphere Model) model to estimate the change in soil carbon stocks for 0–0.3 m depth over time (1990–2008) when a permanent pasture was turned into a kiwifruit orchard (Fig. 10). We used a soil and climate record representative of the main kiwifruit production area in New Zealand around Te Puke.

Under a “bare orchard floor scenario”, the decline was 1.98 kg C m^{-2} (19.8 t C ha^{-1}) in 0–0.3 m depth over 17 years. If pasture were used as a cover crop in the alleys (“50% cover crop scenario”), the decline was 0.76 kg C m^{-2} (7.6 t C ha^{-1}) in 0–0.3 m depth over 17 years. If the entire orchard floor were covered by pasture (“100% cover crop scenario”), we found a carbon increase of about 0.47 kg C m^{-2} (4.7 t C ha^{-1}) soil carbon in 0–0.3 m depth over 17 years.

The SPASMO model, if guided by sound measurements from field studies of biochar incorporation, could be used to model this carbon-change sequence with and without the addition of biochar. Furthermore, if these experiments measured the impact on the performance of the vines and trees, and the environmental services that biochar establishes, we could assess through modelling the benefits, or deleterious consequences, of biochar additions to vineyards and orchards.

Figure 10 Modelled change of soil carbon stocks in the soil of an exemplary kiwifruit orchard with different orchard floor management practices. We used existing records of the climate and soils around Te Puke. The previous land use (before 1990) was permanent pasture. We used permanent pasture as a cover crop.



The uncertainty of these estimates is noted to be high.

Stocks and change for post-2012 mitigation options

To our knowledge, no data sets are available other than those described here from which to derive the potential of different soil carbon management options under horticulture. While we know of one study where the carbon sequestration potential of land management options in Europe was estimated (Smith 2004); that study focused on arable farming, and concluded that the highest carbon sequestration could be achieved by a conversion from cropland to grassland (Table 21). This is another indication that a cover crop on the entire orchard/vineyard floor, and not just in the alleys, would probably be the most promising soil carbon management option if that can be integrated with standard management without causing additional difficulties.

Table 21 Soil carbon (C) sequestration potential of different land management options in Europe. The details of the table are taken from Smith (2004)

Practice	Soil C sequestration potential (t C ha ⁻¹ yr ⁻¹)	Estimated uncertainty
Deep-rooting crops	0.62	>50%
Animal manure	0.38	>50%
Cereal straw	0.69	>50%
Sewage sludge	0.26	>50%
Composting	0.38	>50%
Organic farming	0–0.54	>50%
Convert cropland to grassland	1.2–1.69	>50%

7.5.2 Discussion

Horticulture currently occupies about 70 000 ha. Since 1990, the area used for horticulture has expanded by an average of about 1200 ha per year and at twice that rate since 2000. The dominant crops that have seen most expansion of recent years are apples, kiwifruit and vineyards.

Tate et al. (2005) had estimated that the conversion of grazing land to horticulture would lead to a loss of organic carbon stocks for the 0–0.3 m depth by 9 ± 7 t C ha⁻¹. Only a very small number of New Zealand studies have measured changes in soil carbon. Each study has reported carbon stocks and changes with considerable uncertainty, and the more recent studies provide no consistent evidence to modify the best estimate of soil carbon changes of 9 ± 7 t C ha⁻¹ obtained by Tate et al. (2005).

Horticulture typically involves keeping the ground near the horticultural plants free of other plants both to simplify the task of managing the crops and to reduce the competition for water and nutrients. At the same time, horticulturalists often do not aim for maximised vegetative production as this does not necessarily translate into optimum yields of their economic produce. Both these factors tend to reduce the availability of organic substrate for organic matter formation and are probably the main reasons for the observed losses of soil carbon after conversion to horticulture.

As a consequence, there are likely to be mitigation opportunities through shifting production towards more organic farming, incorporating green waste or manures or allowing an extension of cover crops into the cropped rows. The extent to which such options can be implemented without interfering with normal management practices or reducing economic yields needs to be assessed. There may also be opportunities for using biochar to boost soil carbon, but while some aspects of biochar use are promising, others require further evaluation.

Implications of forecasts and scenarios

There are no specific forecasts of the possible future expansion of the industry, but with the past rate of expansion by 1000–2000 ha per year, horticulture is a very small component of the national carbon budget in any case. In terms of carbon balances, soil losses of an estimated $9 \pm 7 \text{ t C ha}^{-1}$ must also be balanced by likely increases in biomass of a comparable magnitude. The combined carbon change is therefore likely to be very small and possibly positive.

For New Zealand's horticultural products, price premiums, however, are critical, and there are increasing pressures from international protocols (GlobalGap), national standards (PAS 2050, British Standards Institute) and supermarkets (notable Tesco's and Walmart) to eco-verify practices and to certify the carbon footprint. Because the profit of horticultural export products might in the future depend on their carbon footprints and other environmental measures, we need to provide confirmation of the changing stocks of soil carbon in vineyards/orchards.

Effects of information gaps/uncertainties on forecast/scenario reliability

The currently available data are insufficient to answer key questions on soil carbon stocks, their change over time, and the impact of soil carbon management options on other aspects of orchards/vineyard production systems. We need to increase the number of sites sampled, cover other soil orders used by horticulture, and systematically include the range of existing management options. Future work should use standardised sampling depths, archive samples and use global positioning system (GPS) location recordings to facilitate repeat sampling and allow the derivation of trends. Such work should be carried out primarily for New Zealand's 'big three' crops of apples, kiwifruit and grapes, in New Zealand's 'big three' horticultural regions of Hawke's Bay, Bay of Plenty, and Marlborough.

What is the role, the impact and the practicality of the use of horticultural waste recycling to change soil carbon stocks? There is a high rate of wastage of fruit in apple and kiwifruit production because of export quality standards, and there could be useful options to return this unwanted by-product carbon back to the orchard/vineyard. Marc, the crushed grapeskins and pips, provides a waste-stream in viticulture that could likewise be used. There should be a comparison of the cost-benefits and practicality of using such waste as fresh, composted, or biocharred. The mechanisms involved, and the feasibility of the incorporation of biochar into the soils of orchards and vineyards, needs to be carefully tested in the field.

Biophysical modelling of carbon capture by horticultural crops needs to be advanced, and this modelling linked to carbon turnover and fate processes in the soil.

7.6 Conclusions and Recommendations

7.6.1 Present status of studies, datasets, analyses and forecasts

The data currently available are insufficient to answer rigorously key questions on soil carbon stocks, their change over time, and the impact of soil carbon management options in orchards/vineyards.

There are no specific forecasts of the possible future expansion of the industry, but with the past rate of expansion by 1000–2000 ha per year, horticulture is a very small component of the national budget in any case. In terms of carbon balances, soil losses of an estimated $9 \pm 7 \text{ t}$

C ha⁻¹ must also be balanced by likely increases in biomass of a comparable magnitude. The combined carbon change is therefore likely to be very small and possibly even positive.

7.6.2 Key uncertainties, information gaps and research priorities

The role, impact and practicality of the use of horticultural waste recycling to change soil carbon stocks need more research. A comparison of the cost-benefits and practicality of using such waste fresh, composted, or biocharred needs to be carried out. The mechanisms involved, and the feasibility of the incorporation of biochar into the soils of orchards and vineyards, needs to be carefully tested in the field, as it appears both positive and negative effects on production may occur.

To date, sampling has been restricted to a limited number of sites and has incompletely covered the range of existing management options or soil orders used by horticulture. The New Zealand expertise of biophysical modelling of carbon capture by horticultural crops need to be advanced, and this modelling need to be better linked to carbon turnover and fate processes in the soil.

7.6.3 Implications of accounting and mitigation options for New Zealand's post-2012 net position

While the conversion of cropping or pasture to horticulture would normally be regarded as an obvious case of land-use change, establishment of horticulture does not always meet the definition of 'forest'. However, if Article 3.4 of the Kyoto Protocol becomes mandatory, changes in carbon stocks (both soil and vegetation) would be accountable even without a change in land use. Furthermore, if any form of all-lands net-net accounting becomes mandatory after 2012, total carbon changes under horticultural land use would be accountable. Nonetheless, regardless of the post-2012 accounting approach, the contribution to New Zealand's net carbon balance will be very small provided the total area under horticulture remains as small as it is at present.

Changes in carbon stocks may be more significant if forest were to be converted to horticultural land use. While such a land-use change is likely to result in more significant carbon-stock changes, it might not necessarily trigger the definition of a land-use change if the horticultural crop comprised trees that meet the Kyoto Protocol forest definition. In practice, however, very few conversions of existing forest land to horticulture are expected to occur in New Zealand.

While there are a few possible scenarios under which definitional issues could become important in the future, at this stage horticulture is not considered likely to be of more than very minor importance under any post-2012 accounting option, if past land-use conversion trends are maintained.

8. Effects of Erosion on Soil Carbon

John Dymond (Landcare Research), Troy Baisden (GNS)

8.1 Introduction

8.1.1 Scope of study

Globally, soil organic matter holds more carbon than plant biomass and atmospheric CO₂ combined, and can emit or store carbon at a range of timescales—typically extending beyond the 3–5-year timescale of conventional experiments in soil science, agriculture and forestry (Amundson 1991). Much of the report focuses on enhancing information on the role of soils in the New Zealand carbon cycle. Existing New Zealand publications (Tate et al. 2000, 2005; Scott et al. 2006a) emphasise that erosion may represent a proportionally more significant component of the soil carbon cycle in New Zealand than in most other nations. The primary reason for the disproportionate effect of erosion in New Zealand’s carbon cycle is tectonically driven uplift and erosion rates combined with high biological productivity and land-use change (Lyons et al. 2002; Scott et al. 2006a; Hilton 2008a, 2008b). The best available published information on the delivery of particulate C via New Zealand rivers to the ocean is approximately 3 ± 1 Mt C /y, or more than 25% of New Zealand’s fossil fuel C emissions. Despite the large magnitude of this figure, riverine C transport to the ocean may significantly underestimate the true impact of erosion on atmospheric CO₂ by neglecting both the large quantity of eroded sediments that may be deposited on land before reaching the ocean (Stallard 1998), as well as the net effect of erosion, burial and soil recovery on atmospheric CO₂ (Berhe et al. 2007; Van Oost et al. 2007).

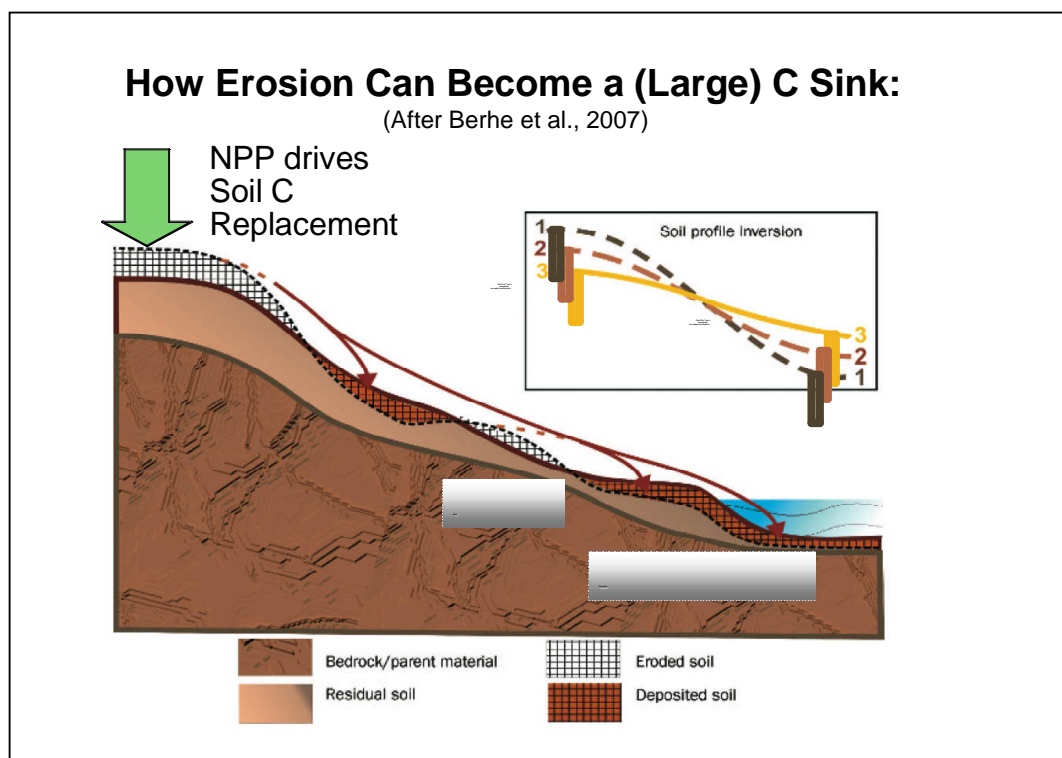
This section provides the first effort to provide a complete estimate of the impact of New Zealand erosion on atmospheric CO₂. As such, preliminary estimates are provided in scientific terms, noting the absence of any current IPCC-endorsed methodology for accounting for the effects of erosion on net national carbon balances. It is apparent that conventional soil carbon-accounting methods employed by the UNFCCC will consider erosion to be a source of C to the atmosphere (e.g., Lal et al. 2003), when more recent publications show that erosion is more likely to be strong or weak carbon sink (Berhe et al. 2007; Van Oost et al. 2007; Galy et al. 2007). The purpose of this material is to provide a basis for considering the potential effect of erosion, inherently included in various methodologies for soil C accounting, on New Zealand’s net position under Kyoto and future agreements, as well as identify research needs to rapidly improve estimates.

8.1.2 Accounting and reporting requirements

Currently, there are essentially no clear accounting and reporting requirements for erosion. The limited guidelines and comments that exist in IPCC documents appear to be in conflict with recent peer-reviewed literature, creating considerable uncertainty in this arena. The IPCC AR4 contains statements such as “Any practice that increases the photosynthetic input of carbon and/or slows the return of stored carbon to CO₂ via respiration, fire or erosion will increase C reserves, thereby ‘sequestering’ carbon or building carbon ‘sinks’”. This statement was incompatible with some elements of the scientific literature when it was written (Stallard 1998; Harden et al. 1999; Manies et al. 2001; Rosenbloom et al. 2001), and is further

questioned by more recent literature (Berhe et al. 2007; Van Oost et al. 2007). The fundamental issue is whether carbon, once eroded, is mainly oxidised to CO₂ or buried and preserved. Current soil C inventory methodologies implicitly assume that eroded C is oxidised to CO₂. The alternative, which is now strongly supported for situations such as New Zealand's (Page et al. 2004; Galy et al. 2007), is that burial and preservation represent a substantial C sink—but a C sink that lies outside of traditional C accounting.

Box 1 The simplified landscape depiction of erosion, transport and deposition, resulting in a dynamic erosion sink. In the inset, traditional soil C inventories are collected at three different times, and the difference in soil C inventory over time is assumed to be source or sink of C to the atmosphere. As shown in the inset, the traditional methodology is not valid if erosion and deposition cause the sampling depths to move vertically over time. The result is generally that eroding land and its associated depositional zones might be considered a CO₂ source under traditional accounting practices, but actually represent a CO₂ sink.



Given the lack of clear statements provided by AR4 or the IPCC GPG for LULUCF (IPCC 2003), it might be acceptable to suggest that accounting for erosion is unnecessary. However, erosion causes errors in soil C accounting (and therefore LULUCF accounting) that can have important consequences. For example, a secondary accounting and reporting issue related to erosion and deposition is the fact that erosion and deposition affect the actual depths of soil (or masses where equivalent mass is used), such that a soil C inventory measured at one time will be measuring a different volume (or mass) before and after erosion or deposition have occurred (see Box 1). A further point is that the inventory method described above and in Box 1 will not match results obtained from verification methods that measure ecosystem level CO₂ exchanges between land and atmosphere, such as eddy covariance. Where a soil C inventory method measures a C loss due to erosion, eddy covariance will observe no CO₂ derived from the ecosystem entering the atmosphere. In fact, neither approach is necessarily correct, because eddy covariance and similar methods will fail

to identify CO₂ entering the atmosphere from areas where sediment is deposited, if the sediment is in fact oxidized in the depositional zone.

These points therefore raise concerns that incorporating erosion in C inventories and accounting could result in considerable complications and inconsistencies—notably where Tier 1 and 2 inventory-based methods might produce a radically different answer than Tier 3 process-based models verified by eddy covariance and other ecosystem-scale CO₂ exchange data. A final set of issues relating to erosion that requires consideration are the following. How can erosion related CO₂ sources and sinks be related to a baseline year or period? And what additionality tests can be applied to changes in erosion and deposition rates resulting from human-induced land-use change?

As a result of the lack of guidance provided by IPCC in the case of erosion, it is beyond the scope of this exercise to provide a range of estimates for different methodologies, accounting, or policy scenarios. However, it is noted that correctly compiled calculations, based on the procedures outlined below, should allow the information and models outlined here to be used to undertake such calculations on a consistent basis.

8.2 The Effect of Erosion on New Zealand Carbon Stocks and Change

8.2.1 Review of New Zealand studies and datasets

In the most complete study of erosion and the C cycle in New Zealand, Scott et al. (2006a) estimated the transport of particulate organic C (POC) to the ocean via New Zealand rivers²⁰ to be $3 \pm 1 \text{ Tg C y}^{-1}$. As noted above, this value is very significant in terms of New Zealand's fossil fuel C emissions, but may underestimate the true impact of erosion on atmospheric CO₂ by neglecting both the large quantity of eroded sediments that may be deposited on land before reaching the ocean (Stallard 1998), as well as the net effect of erosion, burial and soil recovery on atmospheric CO₂ (Stallard 2008; Berhe et al. 2007; Van Oost et al. 2007).

Deposition on land and rates of soil recovery have been estimated based on measurements at a small catchment scale for a landscape typical of highly erodible New Zealand pastoral hill country surrounding Lake Tutira (Page et al. 2004). The landscape budget for Lake Tutira emphasises that typical rates of deposition and soil recovery are important in landslide terrains and may substantially exceed estimates derived from global approaches based on surface erosion models (e.g., Van Oost et al. 2007), which therefore substantially underestimate the net C sink caused by erosion in these landscapes.

The type of sediment and carbon budget developed for Lake Tutira (Page et al. 2004) can be extended to all of New Zealand using relatively simple models and GIS layers developed as part of the programme that produced the national riverine C transport estimates reported in Scott et al. (2006a). The GIS layers constituting “erosion terrains” for New Zealand, and models that use them are detailed below.

²⁰ Some confusion has been observed between the estimate of POC, and the inclusion of Scott et al.'s (2006) estimate of dissolved organic carbon (DOC) transport, to estimate total transport to the ocean of organic carbon derived primarily from soils (dissolved plus particulate) as $4 \pm 1 \text{ Tg C y}^{-1}$. This total estimate of riverine organic C transport is relevant to the use of process-based C cycle models, but the POC estimate of $3 \pm 1 \text{ Tg C y}^{-1}$ remains the best estimate of the transport of eroded C to the ocean via rivers.

8.2.2 Evaluating the effect of erosion on New Zealand's carbon balance

Erosion terrains

Erosion processes vary throughout New Zealand, depending on rock type, landform (especially slope angle) and rainfall. New Zealand was partitioned on the basis of these factors at the scale of 1:50 000 to produce areas with similar erosion processes, called erosion terrains, by amalgamating land use capability units from the New Zealand Land Resource Inventory (Eyles 1983). These terrains were classified, and used as the basis of existing information on New Zealand's erosion C fluxes (Scott et al. 2006a). While differences in land use or management and vegetation cover are important, these were omitted from the definition in order to represent intrinsic erosion susceptibility independently from factors that can change with time. A three-level hierarchical classification was used for both the North and South Islands (Tables A1 and A2 in the Appendix). For the North Island, 9 groups were differentiated at the top level on the basis of landform and slope. At the second level, 26 groups were differentiated on the basis of rock type. At the third level, 52 groups were differentiated on the basis of erosion processes and further detail of rock type. For the South Island, 9 groups were differentiated at the top level based on landform and slope. At the second level, 18 groups were differentiated based on rock type, induration and presence of loess; and at the third level, 37 groups were differentiated on the basis of erosion processes and further detail of rock type.

Erosion models and soil organic carbon content

The carbon loss from a point due to soil erosion is the product of the erosion rate and the soil organic carbon (SOC) content of the soil eroded. The carbon content of soil decreases with depth, so it is necessary to know the depth to which erosion takes place. We use two separate models for this: the New Zealand empirical erosion model (NZeem®) is used to estimate total erosion from all erosion processes; and a New Zealand Universal Soil Loss Equation (NZUSLE) is used to estimate erosion from surficial erosion processes. A third erosion model, the Highly Erodible Land (HEL) model, is used to estimate the sediment delivery ratio for land dominated by landslide erosion. Three national raster layers of SOC (0–10 cm, 10–30 cm, and 30–100 cm) were made by a multiple regression of 1827 measurements of SOC throughout New Zealand on relevant variables existing as national GIS layers. These variables included soil type, land cover, elevation, slope, northing, mean annual rainfall, mean water deficit, mean solar radiation, and mean annual minimum temperature. The r.m.s. errors of the three SOC layers were 0.19, 0.24, and 0.30, respectively.

New Zealand empirical erosion model (NZeem®)

The total erosion rate, $\bar{e}(x, y)$, which varies in space, is estimated from three factors: erosion terrain, mean annual rainfall, and land cover:

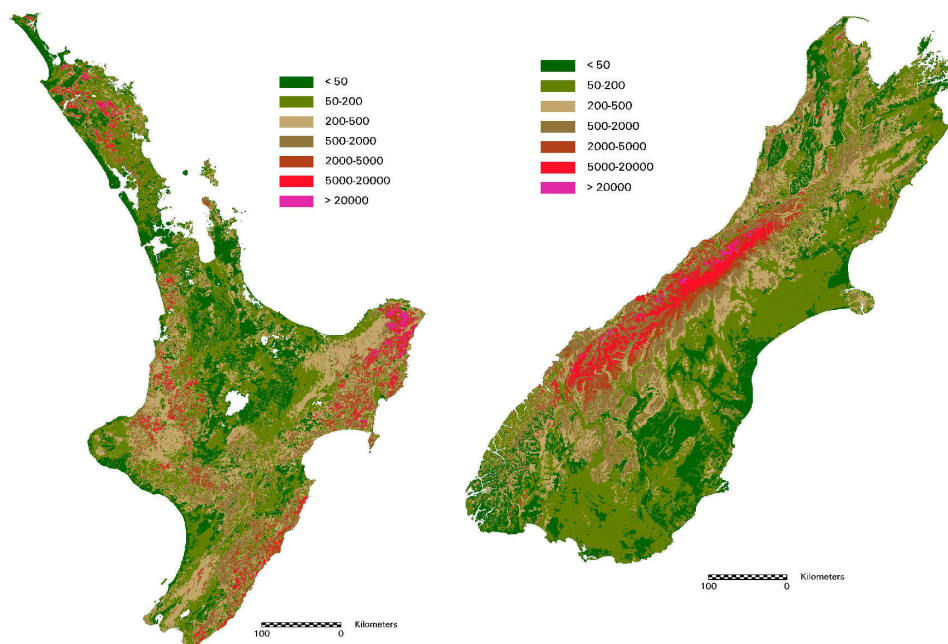
$$\bar{e}(x, y) = a(x, y)C(x, y)P^2(x, y) \quad (1)$$

where $a(x, y)$ is a constant depending on the erosion terrain (termed the erosion coefficient) and is determined by calibration of the model with measurements of long-term sediment yield from 200 sites around New Zealand (Dymond & Betts, submitted); $P(x, y)$ is the mean annual rainfall (mm); and $C(x, y)$ is the erosion rate of the land cover at (x, y) relative to forest. In tectonically active New Zealand, erosion rates are dominated by mass movement erosion. Studies in North Island hill country have shown that when forest is converted to pasture, long-term erosion rates increase by approximately an order of magnitude (Page & Trustrum 1997), as well as erosion rates in major landsliding events (Dymond et al. 2006; Marden &

Rowan 1993). Dymond and Betts (submitted) assumed that $C(x,y)$ could be described for three land-cover types—woody vegetation, herbaceous vegetation, and bare ground—as:

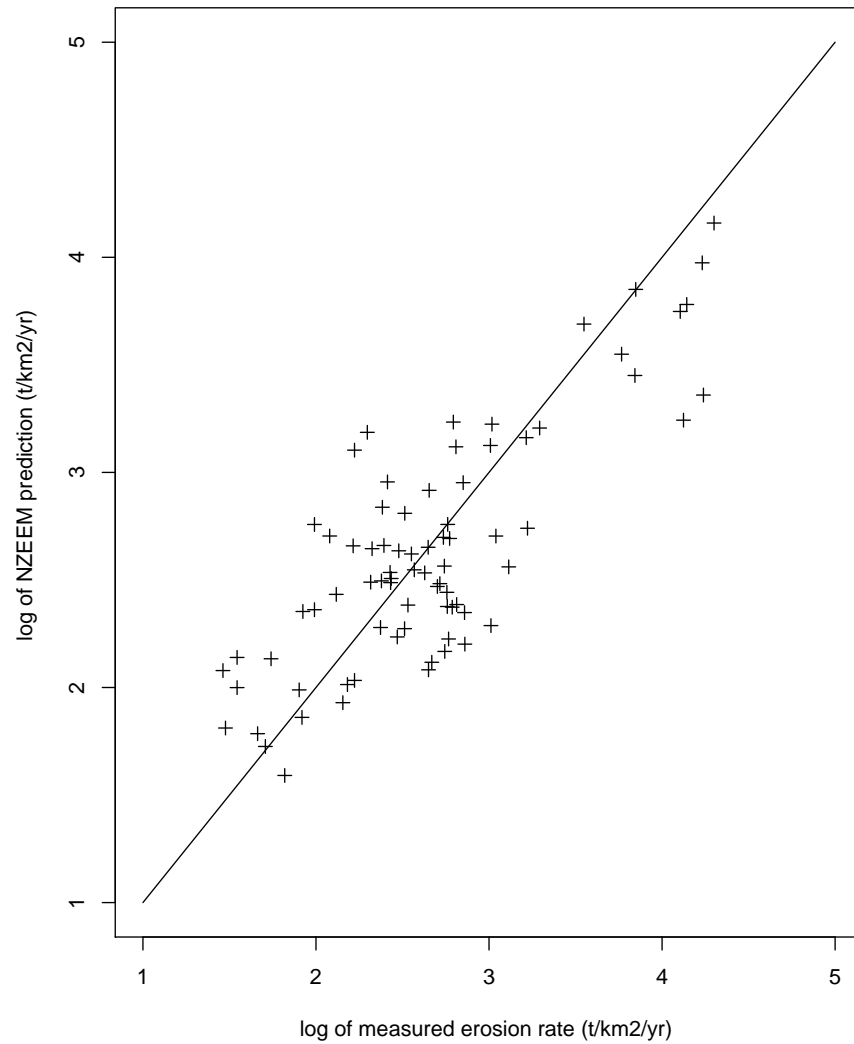
$$\begin{aligned}
 C(x,y) &= 1 \text{ if land cover is woody vegetation} \\
 &= 10 \text{ if land cover is herbaceous vegetation} \\
 &= 10 \text{ if land cover is bare ground.}
 \end{aligned}
 \tag{2}$$

Figure 11 Spatial distribution of erosion rates in the North and South Islands (year 2001) estimated from NZeem®. Source digital data exists on a 15-m grid.



They assigned pasture and bare ground the same cover factor, as neither have deep and strong roots sufficient for strengthening soil to the depth of bedrock: even though bare ground has a much higher surficial erosion rate than herbaceous vegetation, surficial erosion is generally dominated by mass movement erosion (Eyles 1983). A national map of cover factor at 1:50 000 scale (i.e. 15-m pixels) was produced from ETM+ satellite imagery using the method of Dymond and Shepherd (2004). Imagery dates varied between the summers of 1999/2000 and 2002/2003. Figure 11 shows the spatial distribution of erosion rates in New Zealand estimated using NZeem®. The date is nominally 2001, but because land cover has not changed significantly in the last 7 years it may be used to estimate rates in 2008. Figure 12 compares NZeem® predictions with measurements.

Figure 12 Plot of NZeem® predictions versus measured erosion rates for catchments spread throughout New Zealand.



New Zealand Universal Soil Loss Equation (NZUSLE)

A New Zealand version of the Universal Soil Loss Equation was developed for estimating erosion rates from surficial erosion processes (i.e. rill and inter-rill erosion). It has the same factors as the USLE (Wischmeier & Smith 1978), except that the rainfall factor is a function of mean annual rainfall only (following Mitchel & Bubenser 1980). The NZUSLE was calibrated using published data of surficial erosion rates in New Zealand (Basher et al. 1997; Benny & Stephens 1985; Cooper et al. 1992; Dons 1987; Fahey & Coker 1992; Fahey & Marden 2000; Lambert et al. 1985; Mosley 1980; O'Loughlin et al. 1978, 1980; O'Loughlin 1984; Quinn & Stroud 2002; Rodda et al. 2001; Smith 1992; Smith & Fenton 1993; Soons & Rainer 1968; Wilcock 1986; Wilcock et al. 1999). NZUSLE gives the annual erosion rate caused by surficial erosion processes, $\bar{e}_s(x, y)$, as the product of five factors:

$$\bar{e}_s(x, y) = \alpha P^2 KLZU \quad (3)$$

where

$\bar{e}_s(x, y)$ is the erosion rate due to surficial processes (t/km²/yr);

α is a constant calibrated with published surficial erosion rates (1.2×10^{-5});

P is mean annual rainfall (mm / yr);

K is the soil erodibility factor (sand 0.05; silt 0.35; clay 0.20; loam 0.25),

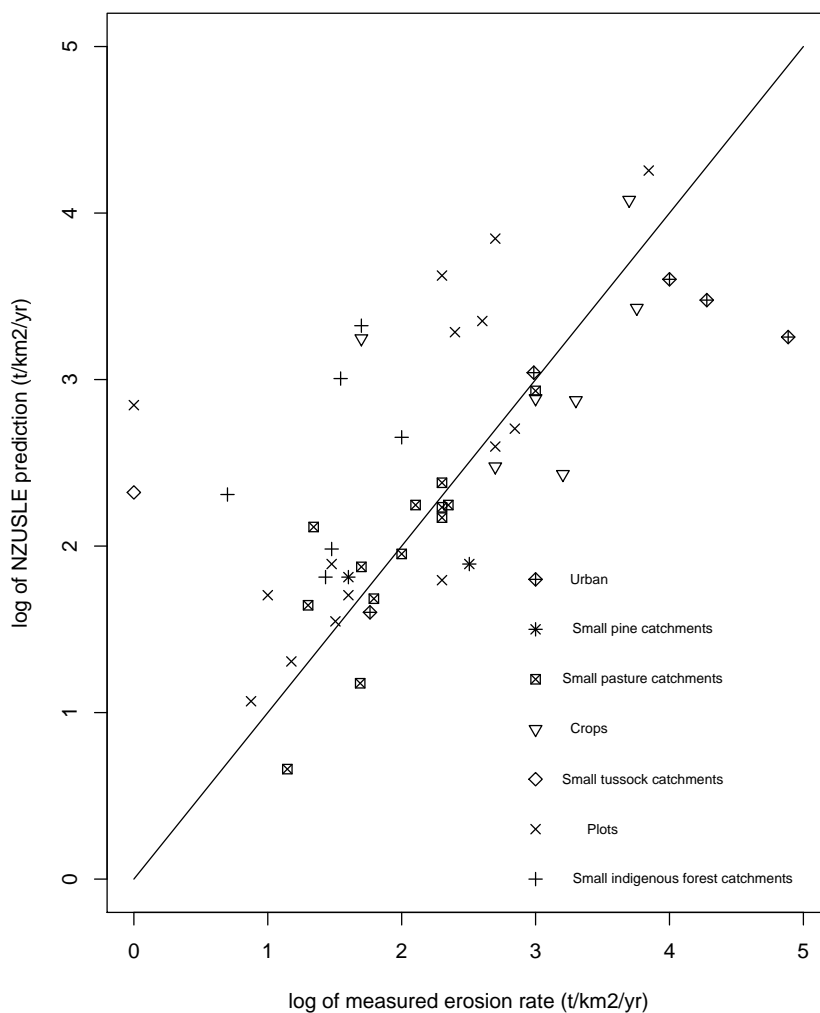
$L = (\lambda / 22)^{0.5}$ where λ is slope length in metres;

$Z = 0.065 + 4.56 \frac{dz}{dx} + 65.41 \left(\frac{dz}{dx}\right)^2$ where dz/dx is the slope gradient (no units); and

U is a vegetation cover factor (bare ground 1.0, pasture 0.01, scrub 0.005, forest 0.005)

Figure 13 compares NZUSLE predictions with the published measurements.

Figure 13 Plot of NZUSLE predictions versus measured rates of surficial erosion spread throughout New Zealand.



Highly Erodible Land Model (HEL)

The HEL model identifies land susceptible to landsliding from three national GIS layers: a land cover map, a slope map from a digital elevation model (DEM) and an erosion terrain map. The GIS layers are rasters with 15-m pixels. For every pixel, the slope is examined to see if it exceeds a threshold set for each rock type (Dymond et al. 2006; Dymond & Betts submitted). If a pixel exceeds the slope threshold and does not have woody vegetation in the land cover map, then it is identified as land susceptible to landsliding. In that case, the flow path down to the nearest stream is traversed in the DEM, using flow direction and flow accumulation, to decide whether the pixel can deliver landslide debris to the stream network. If the flow path encounters any significant flat land, that is, consecutive pixels below four degrees of slope, then the original susceptible pixel is tagged as “non-contributing”, because sediment will deposit on the flat land before it reaches a stream. Otherwise, the pixel is tagged as “contributing”. The proportion of HEL land in an erosion terrain that is tagged as “contributing” is an approximation of the sediment delivery ratio, that is, the proportion of eroded sediment reaching streams.

The effect of erosion on net terrestrial carbon balance

The annual net emission of carbon to the atmosphere from the soil due to erosion processes is denoted by G . It may be estimated by

$$G = f_d D + f_s O - R \quad (4)$$

where:

D is the annual deposition of carbon on the landscape (t/yr) and f_d is the fraction of that emitted to the atmosphere;

O is the annual yield of carbon to the sea (t/yr) and f_s is the fraction of that not buried and eventually released to the atmosphere; and

R is the annual mass of carbon sequestered from the atmosphere in soils regenerating after erosion (t/yr).

New Zealand is a high-standing oceanic island delivering a large mass of sediment to the ocean every year (Lyons et al. 2002). The associated organic carbon will therefore most likely be buried efficiently on the ocean floor with the sediment (Galy et al. 2007; Masiello 2007). Assuming the burial efficiency ranges somewhere between 0.6 and 1.0, we assign a nominal value of 0.2 (± 0.2) to f_s . Similarly, erosion carbon is buried efficiently on the landscape and we assign f_d to zero. So a national budget of G requires the estimation of O and R for each erosion terrain in New Zealand.

Eroded carbon yield to the sea

A raster GIS layer (15-m pixels) of total erosion rate was produced for each erosion terrain using NZeem®. The mean value for the erosion terrain is denoted by S and comprises all erosion processes in the landscape. Because the carbon content of soil increases with depth, it is necessary to apportion the total erosion into surficial (shallow) and mass movement (deep). The NZUSLE was used to estimate mean surficial erosion rates for each erosion terrain and thence the proportion, denoted by p_s .

The annual yield of carbon to the sea, O , from the erosion terrain, may then be estimated by the product of the sediment yield with the mean carbon content of that sediment:

$$O = ((1 - p_s)C_{100} + p_s C_{10})S A_t \quad (5)$$

where

S is the mean erosion rate for the erosion terrain ($\text{t}/\text{km}^2/\text{yr}$);

A_t is the area of the erosion terrain in km^2 ;

C_{100} is the mean carbon content of the top 100 cm of soil for the erosion terrain (no unit);

C_{10} is the mean carbon content of the top 10 cm of soil for the erosion terrain (no unit).

Carbon sequestration in regenerating soils

In the South Island, erosion is dominated by natural processes in the Southern Alps, and there have been no major perturbations of climate or vegetation in the last 5000 years. So, soil erosion and regeneration of soils can be assumed to be approximately in balance (Stallard 1998), that is, $R \approx O$. However, in the North Island, erosion is primarily caused by deforestation in hill country, occurring c.110 years b.p. The regeneration of soils is therefore not necessarily in balance with erosion and needs to be explicitly considered.

There are three processes by which carbon can be sequestered into soils regenerating after erosion:

- (i) shallow landslides remove the soil profile down to bedrock (~ 100 cm) which regenerates back to a normal soil with a depth of ~ 100 cm after about a hundred years;
- (ii) the debris tails of landslides deposit on hillsides, at a depth much shallower than the scar depth and with a carbon content equal to the average of the top 100cm, will sequester carbon until the normal carbon content of the top 10 cm is attained;
- (iii) surficial erosion occurs everywhere and will be approximately balanced by soil regeneration.

On erosion terrains dominated by gully erosion and earthflow, there are assumed to be no landslides and consequently no old landslide scars sequestering carbon. On erosion terrains dominated by landslide erosion, the carbon exported from mass movement erosion is assumed to recover on the old scars at a rate given by an exponential recovery curve (Page et al. 2004; Parfitt, pers. com.):

$$C_s = C_{100}(1 - e^{-0.03t}) \quad (6)$$

where C_s is the carbon content of soil on a landslide scar t years after failure, and C_{100} is the carbon content of the top 100 cm of soil before landslide failure. The mass of carbon sequestered (tonnes) in one year by an area, $a_s(t)$, of scars with age t years (assuming a depth of 1 metre and a bulk density of $1.3 \text{ t}/\text{m}^3$) is then given by:

$$m_s(t) = 1.3 a_s(t) 0.03 C_{100} e^{-0.03t} \quad (7)$$

The total mass of carbon sequestered in one year, M_s , by scars of all ages may be estimated by integrating $m_s(t)$ over time from $t=0$ to $t=T$, the time since deforestation (assumed to be uniformly 100 years for every erosion terrain), that is,

$$M_s = \int_0^T 1.3 a_s(t) 0.03 C_{100} e^{-0.03t} dt \quad (8)$$

The distribution of scar age is assumed to take the form

$$a_s(t) = b e^{-k(T-t)} \quad (9)$$

where b is the yearly area rate of scar production immediately after deforestation, and k is the “coefficient of event resistance”. NZeem® is used to estimate the annual sediment mass reaching streams from landslides, on landsliding erosion terrains, which is adjusted by the sediment delivery ratio (from the HEL model) to estimate b (assuming an average scar depth of 100 cm and bulk density of 1.3). If k is small it may be approximated by the ratio of b over the total area of land available for landsliding, H (from the HEL model). Substituting equation (9) into equation (8) we obtain

$$M_s = 1.3 b C_{100} \frac{0.03}{(0.03 - k)} \{ e^{-kT} - e^{-0.03T} \} \quad (10)$$

The debris tails remaining on the hillside are deposited in a layer thinner (~20 cm) than the ~100 cm depth of landslide scars. The debris tails will have a carbon content of approximately C_{100} and will begin to sequester carbon to achieve a carbon content of C_{10} at a rate similar to the bare erosion scars (the rate is controlled primarily by the phosphorus content). Assuming the debris tails are not buried by subsequent landsliding, the mass of carbon sequestered by landslide debris in a year, M_d , will be approximately the sediment delivery ratio, β , times the carbon sequestered by old landslide scars, that is,

$$M_d = \beta M_s \quad (11)$$

Surficial erosion is generally small in comparison with mass-movement erosion, and it is assumed that soils undergoing surficial erosion are sequestering carbon at a rate equal to the erosion carbon loss rate. The mass of carbon sequestered in a year by soils undergoing surficial erosion, M_u , is then given by:

$$M_u = p_s O \quad (12)$$

Results for contemporary New Zealand

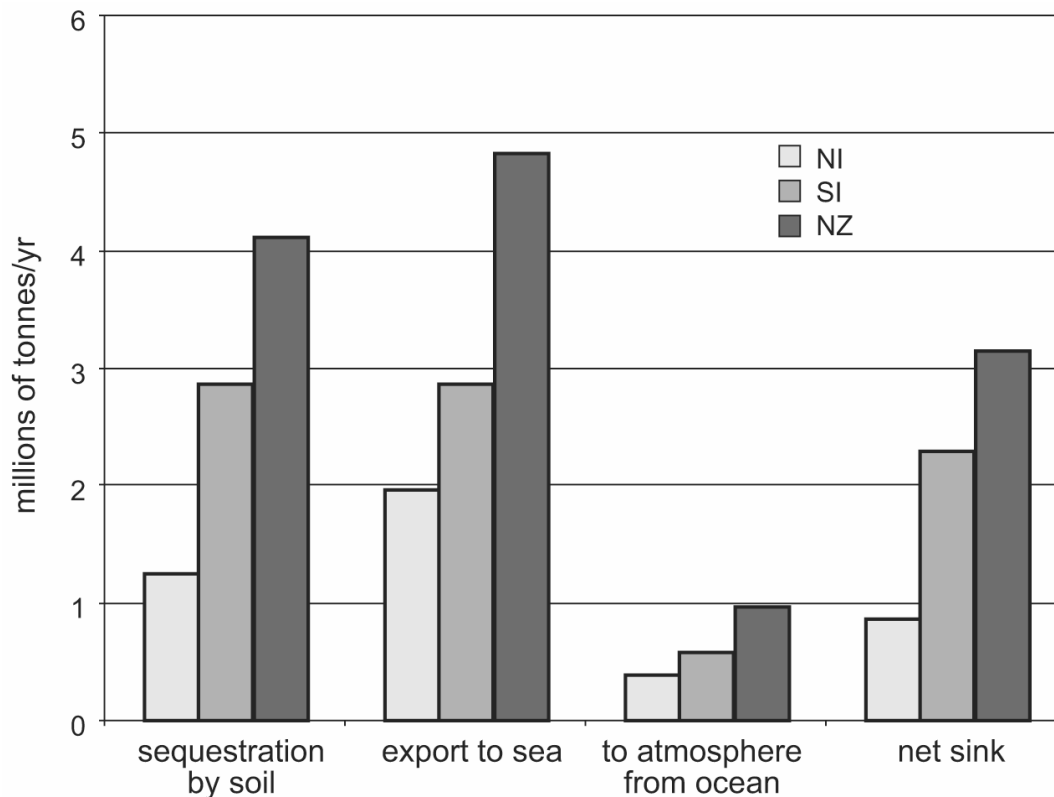
The following results describe an erosion carbon budget for contemporary New Zealand (e.g., 2008). The carbon export to the ocean (due to erosion) for the North Island is estimated to be 1.9 Mt C/ yr (Table A1, Appendix). There are 6 erosion terrains that contribute more than 0.10 Mt C/ yr to this total: the two earthflow terrains, 632 and 633, export 0.25 and 0.24 Mt C/ yr, respectively; the two gully terrains, 634 and 732, export 0.29 and 0.41 Mt C/ yr; the hill country on mudstone terrain, 731, exports 0.13 Mt C/ yr; and hill country on sandstone, 741, exports 0.10 Mt C/ yr. The carbon export to the ocean (due to erosion) for the South Island is estimated to be 2.9 Mt C/ yr (Table A2, Appendix), 1 Mt C more than from the North Island. There are 6 erosion terrains that contribute more than 0.10 Mt C/ yr to the

South Island total: active flood plains, 111, export 0.28 Mt C/ yr; non-loess terraces and fans, 411, export 0.10 Mt C/ yr; hilly steeplands in soft-sandstone, 712, export 0.13 Mt C/ yr; mountain steeplands in hard sedimentary rocks, 811, export, 0.50 Mt C/ yr; mountain steeplands in schist rocks, 812, export 1.40 Mt C/ yr; and alpine slopes, 9, export 0.14 Mt C/ yr. The North and South Islands together export 4.8 Mt C/ yr of carbon per year to the ocean.

The sequestration of carbon by soils in the South Island is assumed to be in balance with the carbon export by erosion, that is, 2.9 Mt C/ yr. The contribution of individual erosion terrains to this total is the same as their contribution to carbon export. In the North Island, the sequestration of carbon by soils is estimated to be 1.25 Mt C/ yr, which comprises 0.65 Mt C/ yr from landslide scars, 0.3 Mt C/ yr from debris tails, and 0.3 Mt C/ yr from surficial erosion sites. The total of 1.25 Mt C/ yr is significantly less than the 1.9 Mt C/ yr being exported to the ocean, so there is a current net loss of carbon from North Island soils (due to erosion) of approximately 0.65 Mt C/ yr. This net loss is occurring primarily in the earthflow and gully terrains where there is negligible sequestration of carbon by soils. On the erosion terrains where landsliding is the dominant erosion process, there is a net increase of soil carbon: for example, the hill country on mudstone terrain, 731, is sequestering 0.29 Mt C/ yr, which is over twice the export.

Figure 14 shows the net sink of carbon for the North and South Islands, and New Zealand in total. There is a net sink of 0.85 Mt C/ yr for the North Island, a net sink of 2.30 Mt C/ yr for the South Island, making a total of 3.15 Mt C/ yr for New Zealand. The contribution of individual erosion terrains to the net carbon sink in the South Island follows a similar pattern to that for soil sequestration of carbon: that is, active flood plains, 111, sink 0.22 Mt C/ yr; hilly steeplands in soft-sandstone, 712, sink 0.10 Mt C/ yr; mountain steeplands in hard sedimentary rocks, 811, sink, 0.40 Mt C/ yr; mountain steeplands in schist rocks, 812, sink 1.12 Mt C/ yr; and alpine slopes, 9, sink 0.10 Mt C/ yr. In the North Island, there are three erosion terrains that contribute more than 0.10 Mt C/ yr to the net sink: the hill country on mudstone terrain, 731, sinks 0.26 Mt C/ yr; hill country on sandstone, 741, sinks 0.18 Mt C/ yr; and hill country on greywacke, 761, sinks 0.10 Mt C/ yr.

Figure 14 Annual carbon transfers associated with soil erosion for North Island, South Island, and New Zealand at 2008.



8.2.3 Results in the context of changing international views of erosion and the carbon cycle

It is now well described in the scientific literature that erosion plays a disproportionately large role in the carbon cycle of New Zealand, where active mountain belts combine tectonically driven uplift and erosion rates with high biological productivity and land-use change (Lyons et al. 2002; Scott et al. 2006a; Hilton et al. 2008a, 2008b). The best available published information on the delivery of particulate C via New Zealand rivers to the ocean is approximately 3 ± 1 Mt C/ yr, or more than 25% of New Zealand's fossil fuel C emissions. Despite the magnitude of this figure, riverine C transport to the ocean may significantly underestimate the true impact of erosion on atmospheric CO₂ by neglecting both the large quantity of eroded sediments that may be deposited on land before reaching the ocean (Stallard 1998), as well as the net effect of erosion, burial and soil recovery on atmospheric CO₂ (Stallard 2008; Berhe et al. 2007; Van Oost et al. 2007).

The section above provides the first effort to provide a complete estimate of the impact of New Zealand erosion on atmospheric CO₂. As such, preliminary estimates are provided in scientific terms, noting the absence of any current IPCC-endorsed methods for accounting for the effects of erosion on net national carbon balances or accounting. It is apparent that conventional soil carbon accounting methods employed by the UNFCCC will consider erosion to be a source of C to the atmosphere (e.g., Lal et al. 2003), when more recent publications show that erosion is more likely to be strong or weak carbon sink (Berhe et al. 2007; Van Oost et al. 2007; Galy et al. 2007).

While leading international science endeavours to estimate the magnitude of the likely erosion sink (Berhe et al. 2007; Van Oost et al. 2007), it is important to note that much of the apparent net sink indicated by the calculations presented here is a result of entirely natural processes in landscapes with negligible human-induced land-use change (Hilton et al. 2008a, 2008b; Lyons et al. 2002). Despite this, changes in sedimentation resulting from land-use change in many areas of New Zealand are known, and are associated with significant C transport rates (Page et al. 2004; Scott et al. 2006a). Thus, while efforts to identify the magnitude of erosion's influence on the C cycle have received some international attention, there appears to be no focus on identifying the role of erosion and deposition on the carbon cycle in ~5 year increments expected under C accounting agreements.

Furthermore, there is no basis for determining whether any net erosion-induced net sink or source is human-induced or natural, nor is there any identified process for determining whether an erosion-induced net sink or source is additional to that which would have occurred in the absence of a policy or before a baseline year. Instead, international efforts are aimed largely at simply elucidating the degree to which our present knowledge of erosion and deposition implies that the IPCC methodology incorrectly assumes that erosion can only represent a source of C to the atmosphere. IPCC approaches are summarised in AR4 by the statement, "Any practice that increases the photosynthetic input of carbon and/or slows the return of stored carbon to CO₂ via respiration, fire or erosion will increase C reserves, thereby 'sequestering' carbon or building carbon 'sinks'". As a result, the most important initial objective of preliminary accounts for the role of erosion in the C cycle should be to identify the potential for significant errors and perverse outcomes in soil C accounting.

8.3 Erosion and Carbon Mitigation Opportunities

While the modelling presented above does not directly examine specific mitigation opportunities, it is possible to briefly identify and examine some major opportunities. Before examining individual opportunities, it is important to emphasise that the new understanding of erosion as a potential net sink requires a change in the thinking about erosion mitigation. The key change is that the maximum C mitigation will be achieved not simply by reducing erosion, but by maintaining plant production while allowing some erosion to occur. This concept is described in Box 1. In principle, it implies that focus should be placed on maximizing the "greenness" of eroding landscapes, rather than on eliminating or minimising erosion altogether. These concepts can be examined in the context of some specific options related to other potential C mitigation policies.

8.3.1 Assessment of options

Afforestation/reforestation

The principal benefit of afforestation/reforestation can generally be expected to be the sequestration of C in biomass, and in some cases the production of a carbon-neutral fuel or product. Soil C changes are relatively small compared to biomass C accumulation. Despite this, under some circumstances where pastoral stepland remains highly productive despite ongoing erosion, it is possible that the replacement of pasture with forest cover may reduce erosion that provides a long-term C sink. In this sense, a trade-off might exist between a relatively short term C sequestration benefit through biomass C accumulation (~25–50 years) and a much smaller but ongoing C sink due to erosion and deposition. It is reasonable to expect that such a trade-off might occur in landslide prone New Zealand hill country (e.g.,

Page et al. 2004). Many other areas exist where earthflows and particularly gullies continue to expand, generating unproductive areas without substantial plant cover. In these cases, preventing gully expansion and mitigating areas already damaged by erosion are practices that remain win-win in terms of both biomass and soil carbon after including erosion-related C transfers. The calculations presented above (Table A3, Appendix) suggest the apparent net sink activity of landslide terrains (e.g. 631 and 731) may be large relative to gully and earthflow dominated terrains (e.g. 632 and 732). Therefore, the potential sink implications of long-term pastoral land use versus afforestation/reforestation deserve further investigation. An important consideration in further investigations should be the intergenerational equity issue imposed by “turning off” a small net sink, especially on Maori land.

Deforestation

The considerations involved in deforestation are largely those described above for afforestation, but in reverse. Thus, the large net-release of C from forest clearing may, under some circumstances, be replaced by a small net sink induced by erosion and deposition under productive land use. The magnitude of the effect and degree to which a net sink may occur will both be dependent upon the terrain type.

Grazing land

Pasture land remains New Zealand’s dominant productive land-use and remains prominent on large areas of hill country. It has become clear that oversowing with legumes and superphosphate have the potential to restore much of the pasture production and soil carbon in sites where the upper meter of soil has been lost to landslides (Sparling et al. 2003; Page et al. 2004). When combined with the likely burial, and protection from decomposition, of soil C in the debris tail of the landslide, these areas have significant net sink potentials. In addition, surficial erosion processes which are insignificant in terms of sediment, may provide as much as half the total eroded C in landscapes dominated by landslides (Page et al. 2004). As a result, it is apparent that the combination of erosion and deposition associated with ordinary sheetwash erosion may contribute a substantial amount of C within the terrestrial landscape. If it were possible to include erosion within a proper soil C accounting framework, it appears that the greatest gains could be made by enhancing the productivity of the landscape, rather than reducing (or enhancing erosion). For accounting to be useful, there would have to be a baseline against which additional activities can be compared. An example of this might be undertaking revegetation that would otherwise not be economic. Revegetation activities could potentially be part of a complex set of climate change feedbacks, for instance maintaining production in the face of more frequent droughts and/or storms associated with climate change as areas become more marginal for farming.

Cropland

Cropland erosion has generally not been specifically examined in our calculations due to the relatively small area of cropland in New Zealand. Despite this, and due to the fact that cropland tends to occupy relatively flat portions of the landscape, several mitigation opportunities can be noted. First, no-tillage and minimum tillage practices can substantially reduce erosion and improve soil structure and properties in croplands. Second, croplands may have fallow (no plant cover) periods that enhance erosion risk and have low or no plant production. These factors tend to reduce soil carbon. Third, croplands can still represent a net sink where deposition and preservation are significant (Berhe et al. 2007; van Oost 2007). Thus, the combination of tillage practices, plant cover, and depositional zone management may all have the potential to create mitigation opportunities. In the case of croplands, additional advantages include a wealth of overseas research, and the clear case that changed

management has directly led to observed levels of C mitigation. A final point about croplands in New Zealand is that many pastures go through a limited cropping phase during pasture renewal, during which the principles of cropland mitigation may apply.

8.3.2 Environmental co-benefits and risks

The main co-benefits associated with traditional erosion reduction practices are the reduction in sediment and nutrient pollution of surface water. These benefits are less clear if advice is redesigned to maximise the “greenness” of a landscape still experiencing sustained rates of erosion. In fact, maximising “greenness” may imply maintaining a high level of fertility that will lead to a high nutrient content in runoff. Despite this, where erosion and deposition represent a C sink, these activities may be recognised as a co-benefit of what is often the most economically productive land use—pastoral farming.

Land-based mitigation often faces substantial risk from events such as fires and storms. In contrast to the threat of fires, storms, insects or other disturbances to forests, it is generally expected that disturbance can enhance the C sink associated with landscapes already acting as net C sink via erosion and deposition. In other words, a large storm would be likely to mobilise considerable soil in landslides and sheetwash, that would be deposited downslope and onto the seafloor. Providing that the rate of soil C recovery on the eroded areas exceeds the rate of oxidation of C in deposited sediments and buried soils, then the storm event will act as a net C sink. Thus, there appears to be a substantial contrast in storm-induced risk to net C emissions when comparing erosion versus afforestation/reforestation.

8.4 Current Best Estimates of the Effect of Erosion on Carbon Stocks

Stocks and change under current land use/management (1990–2020)

Our current best estimate of an erosion carbon budget for contemporary New Zealand (e.g., ~2008) was reported above in section 4.2.5.2 and the key elements for understanding the effect of erosion on soil C stocks are summarised here. The carbon export to the ocean (due to erosion) for the 1.9 Mt C/ yr from the North Island, (Table A3) and 2.9 Mt C/ yr from the South Island (Table A4). The sequestration of carbon by soils in the South Island is assumed to be in balance with the carbon export by erosion, that is, 2.9 Mt C/ yr. In the North Island, the sequestration of carbon by soils is estimated to be 1.25 Mt C/ yr, which comprises 0.65 Mt C/ yr from landslide scars, 0.3 Mt C/ yr from debris tails, and 0.3 Mt C/ yr from surficial erosion sites. The total of 1.25 Mt C/ yr is significantly less than the 1.9 Mt C/ yr being exported to the ocean, so there is a current net loss of carbon from North Island soils (due to erosion) of approximately 0.65 Mt C/ yr. This net loss is occurring primarily in the earthflow and gully terrains where there is negligible sequestration of carbon by soil recovery after erosion. On the erosion terrains where landsliding is the dominant erosion process, there is a net increase of soil carbon due to recovery: for example, the hill country on mudstone terrain, 731, is calculated to be sequestering 0.29 Mt C/ yr—over twice the export. Thus, the effect of erosion on soil C stocks can be summarised as depending on the terrain type, and whether the landscape is in equilibrium with respect to erosion. Overall, we calculate that New Zealand’s observable soil C stocks are being reduced by erosion under contemporary land management.

Full C accounting should include the fate of eroded soil C. After accounting for preservation after deposition, both islands appear to be net sinks (Fig. 14). There is a net sink of 0.85 Mt C/ yr for the North Island, a net sink of 2.30 Mt C/ yr for the South Island, making a total of 3.15 Mt C/ yr for New Zealand. Although these net sink calculations represent our most

complete estimates to date, it is important to note again that these remain preliminary estimates and that there is no IPCC endorsed framework under which we can account for the fluxes of C related to erosion and deposition, or make adjustments to soil C inventory related to erosion and deposition.

Stocks and change for post-2012 mitigation options

In the case of erosion and deposition, the primary issue post-2012 remains identifying a framework in which erosion might be sensibly accounted for. Such an accounting framework would need to provide an inventory of erosion and deposition fluxes of sediment and carbon, in the form of baseline estimates and changes associated with mitigation activities that may be considered additional to business as usual. Until such a framework is in place, with known guidelines and supporting science capable of discerning net C sources and sinks on ~5-year time-scales, stocks and changes associated with erosion cannot be usefully estimated for mitigation actions.

Furthermore, our framework does not currently provide an estimate for any mitigation options. The most useful information our calculations can provide in the context of the potential for post-2012 mitigation options is the approximate magnitude of the human-induced element of the erosion C budget, and the degree to which it can be modified by incentives for further mitigation. The simplest approach to estimating the human-induced component of the budget is to simply assume that the North Island is dominated by human activity, while the South Island is essentially in a natural state of erosion. This is consistent with the often quoted observation that North Island erosion rates have increased by a factor of ~5 since European settlement. If we assume the net sink could be enhanced on the order of 10–30% by decreasing erosion in gully terrains and enhancing plant productivity on former landslides and eroding hillslopes, then there would seem to be potential for 0.085–0.25 Mt C/yr of additional C sink activity on the North Island.

8.5 Conclusions and Recommendations

Present status of studies, datasets, analyses and forecasts

Present studies and information are limited in both their completeness and accuracy. This work contains the most current analyses and suggests that erosion and deposition on the New Zealand landscape cause an ongoing C sink, with significant variations in the magnitude of this sink resulting from human-induced land-use changes. We currently estimate that there is a net sink of 0.85 Mt C/yr for the North Island, and a net sink of 2.30 Mt C/yr for the South Island, making a total of 3.15 Mt C/yr for New Zealand. Forecasts for post-2012 are not feasible because rules are largely unknown. Therefore focus must be on gaps, uncertainties and research priorities, in the sense of determining what could be included under future international agreements, and when this might be feasible.

Key information gaps, uncertainties, and research priorities

Major information gaps and uncertainty remain, and can be separated into two categories. Uncertainty in the net effect of erosion and deposition on New Zealand's carbon balance results mainly from the following sources:

- Understanding of the fate of deposited C—how much is oxidised to CO₂?
- The proportion of eroded C that was derived from ancient sources in rocks (Gomez et al. 2003; Hilton et al. 2008a, 2008b, Galy et al. 2008), rather than from the atmosphere-biosphere system.

- Understanding the recovery rate of soil C stocks from erosion.
- Understanding the amount of eroded C that is deposited on land, on downslope soils (colluvium) and floodplains (alluvium).

Uncertainty in the changes in erosional and depositional carbon fluxes due to current (e.g., post-1990) and potential future human-induced land-use changes can be associated with the following sources.

- Sediment and carbon yields from current and proposed future land use and management systems.
- The degree to which land use and terrain type influence the fate of generated sediment.
- The degree to which the bioavailability of sediment from different land uses influences the rate at which sedimentary carbon is returned to the atmosphere from zones of deposition.

Collectively, these sources of uncertainty should represent future research priorities.

8.5.1 Implications of accounting and mitigation options for New Zealand's post-2012 net position

Erosion could have a large impact on New Zealand's post-2012 net position under some circumstances, including circumstances where it is inadvertently included. However, at this stage the magnitude of these effects remains difficult to estimate due to both scientific uncertainty and the lack of an agreed international carbon accounting framework for erosion. As a result, the following positions are suggested. Since the most logical course is for erosion and deposition to be considered by UNFCCC reporting for a substantial period before inclusion in C accounting occurs, the best estimate for post-2012 is a nil effect on New Zealand's net position, for at least 5–10 years. In an optimistic scenario, in which mitigation options for reducing erosion and enhancing rates of soil C recovery following erosion were aggressively pursued in the North Island, an additional sink of up to 0.25 Mt C/ yr appears plausible. In a pessimistic scenario where erosion's perverse impact on soil C accounting (see Box 1) was inadvertently included in a post-2012 agreement, it is plausible that the 0.85 Mt C/ yr sink we currently estimate for the North Island could be accounted for a source of C to the atmosphere.

Appendix - Details of the Erosion Modelling

Table A1 Description of North Island erosion terrains

Label	Description
	1. Active flood plains
1.1.1	Undifferentiated alluvium from modern overbank depositional events. Parts may be Peaty. Includes non-peaty wetlands.
	Sand country
2.1.1	Recent fresh dune sand.
2.1.2	Mature moderately weathered dune sand.
	2. Peatland
3.1.1	Organic soils on deep peat.
	Terraces, low fans, laharc aprons (most slopes <8°)
4.1.1	Loess
4.1.2	Young tephra, mostly pumiceous (waimihia and younger).
4.1.3	Basins infilled with taupo tephra flow deposits—intensely gullied.
4.1.4	Mid-aged (late pleistocene/early holocene) tephra, older tephra, or tephric loess.
4.2.1	Fine grained, weathered, undifferentiated terrace alluvium—above the level of modern Flood plains.
4.3.1	Gravelly soils on alluvial terrace gravels or on gravelly laharc aprons—above the level of modern flood plains.
	Downland (most slopes 8–15°)
5.1.1	Loess
5.1.2	Young tephra (waimihia and younger), over older tephra.
5.1.3	Mid-aged (late pleistocene/early holocene) tephra, older tephra, or tephric loess.
5.2.1	Young basalt lava fields and low domes (parts are flatter than typical downland).
5.3.1	Weathered sedimentary and non-tephric igneous rocks.
	3. Hill country (most slopes 16–25°)
6.1.1	Loess
6.1.2	Young tephra (waimihia or younger), usually over older tephra—shallow (0.3–1.0 m).
6.1.3	Young tephra (waimihia or younger), usually over older tephra—deep (>1.0 m).
6.1.4	Mid-aged (late pleistocene/early holocene) tephra, or tephric loess.
6.2.1	Relatively young basalt domes and cones.
6.3.1	Weak to very weak tertiary-aged mudstone.
6.3.2	Crushed tertiary-aged mudstone, sandstone; argillite, or ancient volcanic rock (frequently, with tephra covers in the northern Hawke’s Bay–East Coast area)—with moderate earthflow-dominated erosion.
6.3.3	Crushed mudstone or argillite with severe earthflow-dominated erosion.
6.3.4	Crushed argillite, sandstone, or greywacke, with severe gully-dominated erosion.
6.4.1	Cohesive, generally weak to moderately strong tertiary-aged sandstone.
6.4.2	Non-cohesive tertiary-aged sandstone.
6.5.1	Limestone
6.6.1	Unweathered to moderately weathered greywacke/argillite.
6.6.2	Unweathered to slightly weathered white argillite.
6.7.1	Residual weathered to highly (often deeply) weathered tertiary-aged sedimentary rocks.
6.7.2	Residual weathered to highly (often deeply) weathered ancient basalt and andesite.
6.7.3	Residual weathered to highly (often deeply) weathered welded rhyolite.
6.7.4	Residual weathered to highly (often deeply) weathered greywacke/argillite.

Hilly steeplands (most slopes >25°)

- 7.1.1 Young tephra (waimihia or younger), usually over older tephra—shallow (0.3–1.0 m) covers.
- 7.1.2 Young tephra (waimihia or younger), usually over older tephra—deep (>1.0 m).
- 7.1.3 Mid-aged (late pleistocene/early holocene) tephra.
- 7.2.1 Fresh to slightly weathered welded rhyolitic rock, or bouldery, andesitic lahar deposits.
- 7.3.1 Weak to very weak tertiary-aged mudstone.
- 7.3.2 Crushed argillite with gully-dominated erosion.
- 7.4.1 Cohesive, generally weak to moderately strong tertiary-aged sandstone.
- 7.4.2 Non-cohesive tertiary-aged sandstone, and younger sandy gravels and gravelly sands.
- 7.5.1 Limestone
- 7.6.1 Unweathered to moderately weathered greywacke/argillite.
- 7.6.2 Unweathered to slightly weathered white argillite.
- 7.7.1 Residual weathered to highly (often deeply) weathered ancient basalt and andesite.
- 7.7.2 Residual weathered to highly (often deeply) weathered welded rhyolite.
- 7.7.3 Residual weathered to highly (often deeply) weathered greywacke/argillite.

4. Upland plains and plateaux

- 8.1.1 Upland plains and plateaux with tephra covers.

5. Mountain steeplands

- 9.1.1 Greywacke/argillite or younger sedimentary rocks of the main ranges prone to landslide erosion.
 - 9.1.2 Greywacke/argillite or younger sedimentary rocks of the main ranges prone to sheet/wind/scree erosion.
 - 9.2.1 Volcanic rocks in mountain terrains and upland hills.
 - 9.2.2 Upper flanks of volcanoes.
-

Table A2 Description of South Island erosion terrains

Label	Description
	Active flood plains
1.1.1	Recent (young), active floodplains and fans flat to gently sloping.
	Sand country
2.1.1	Coastal sand dunes, beach ridges, flat to moderately sloping sand flats, sand dunes.
	Peatland
3.1.1	Peat deposits flat to gently undulating peat swamps, domed and upland peat deposits.
	Terraces and fans
4.1.1	Flat to gently sloping terraces and fans of older alluvium above the floodplain.
4.2.1	Loess on flat to gently sloping terraces and fans of older alluvium above recent floodplain.
	Downland (most slopes 4–15 degrees)
5.1.1	Moraine and dissected alluvium.
5.2.1	Loess > 1m deep.
5.3.1	Soft sedimentary rocks.
5.4.1	Hard sedimentary rocks.
5.4.2	Hard schist rocks.
5.4.3	Hard coarse-grained igneous or metamorphic and fine igneous rocks.
	Hill country (most slopes 16–25 degrees)
6.1.1	Moraine or dissected alluvium.
6.2.1	Loess > 1m deep.
6.3.1	Soft sedimentary mudstone.
6.3.2	Soft sedimentary sandstone.
6.3.3	Soft sedimentary conglomerate.
6.3.4	Soft calcareous sediments.
6.4.1	Hard sedimentary rocks.
6.4.2	Hard schist rocks.
6.4.3	Hard coarse-grained igneous or metamorphic rocks.
6.4.4	Hard fine grained igneous rocks.
	Hilly steplands (most slopes > 25 degrees)
7.1.1	Soft mudstone.
7.1.2	Soft sandstone.
7.1.3	Soft conglomerate.
7.2.1	Hard sedimentary rocks.
7.2.2	Hard schist rocks.
7.2.3	Hard coarse-grained igneous or metamorphic rocks.
7.2.4	Hard carbonate rocks.
7.2.5	Fine grained igneous rocks.
7.3.1	Weathered hard schist & greywacke rocks.
7.3.2	Weathered coarse-grained igneous rocks.
	Mountain steplands
8.1.1	Hard sedimentary rocks.
8.1.2	Hard schist rocks.
8.1.3	Hard coarse-grained igneous and metamorphic rocks.
8.1.4	Hard fine-grained igneous.
8.1.5	Weathered coarse-grained igneous rocks.
9	Alpine slopes – very steep to precipitous mountain slopes.
	Other land
10	Ice and snow
11, 12	Other

Table A3 Annual carbon transfers associated with soil erosion for erosion terrains in the North Island (export, soil sequestration, release from ocean, net emission). Description of erosion terrains are given in Table A1

Erosion terrain	Area (ha)	Carbon export to sea due to soil erosion (t/yr)	Sequestration of carbon by soil regenerating from erosion (t/yr)	Release of erosion carbon to atmosphere from sea (t/yr)	Net emission of carbon to atmosphere due to erosion (t/yr)
111	833 751	31 793	0	6359	6359
211	199 198	1828	1810	366	-1444
212	125 765	1970	1950	394	-1556
311	147 468	1828	1810	366	-1444
411	90 360	1700	1683	340	-1343
412	277 246	14 436	14 292	2887	-11 404
413	42 039	2792	2764	558	-2206
414	369 955	13 150	13 018	2630	-10 388
421	9440	730	723	146	-577
431	196 497	11 071	10 960	2214	-8746
511	76 610	509	300	102	-198
512	310 949	5572	3118	1114	-2004
513	653 717	13 883	7603	2777	-4826
521	85 954	2113	1161	423	-738
531	376 231	5914	3542	1183	-2360
611	112 767	982	1899	196	-1702
612	459 925	27 589	52 805	5518	-47 288
613	256 388	8174	6700	1635	-5065
614	626 447	27 928	62 518	5586	-56 932
621	15 612	231	1367	46	-1321
631	435 301	36 668	89 279	7334	-81 946
632	361 071	247 271	0	49 454	49 454
633	155 512	239 799	0	47 960	47 960
634	71 616	287 958	0	57 592	57 592
641	413 913	44 268	108 566	8854	-99 712
642	59 375	9659	27 911	1932	-25 979
651	102 564	1587	2617	317	-2300
661	260 849	17 225	39 817	3445	-36 372
662	38 828	504	86	101	15
671	161 143	2618	7506	524	-6982
672	192 006	2906	5649	581	-5068
673	24 446	410	1092	82	-1010
674	200 495	7827	32 695	1565	-31 130
711	88 210	5689	11 246	1138	-10 108
712	106 163	5481	5426	1096	-4330
713	126 675	4423	4379	885	-3494
721	51 039	2239	2216	448	-1769
731	369 297	130 890	290 784	26 178	-264 606
732	56 271	413 901	0	82 780	82 780
741	773 873	100739	203 286	20 148	-183 138
742	26 314	6741	10 490	1348	-9142
751	10 743	1569	356	314	-43
761	581 100	67 789	115 847	13 558	-102 289
762	27 898	809	1843	162	-1681
771	122 158	3109	4401	622	-3779
772	29 703	568	982	114	-869
773	32 779	1364	3408	273	-3135
811	74 196	1113	499	223	-276
911	618 150	97 215	63 954	19 443	-44 511
912	204 252	35 351	22 764	7070	-15 694
921	99 868	1900	1881	380	-1501
922	33 700	1222	1210	244	-966
Total	11 175 827	1 958 593	1 250 239	391 719	--858 521

Table A4 Annual carbon transfers associated with soil erosion for erosion terrains in the South Island (export, soil sequestration, release from ocean, net sink). Description of erosion terrains are given in Table A2

Erosion terrain	Area (ha)	Carbon export to sea due to soil erosion (t/yr)	Sequestration by soil regenerating from erosion (t/yr)	Release of erosion carbon to atmosphere from sea (t/yr)	Net carbon yield to atmosphere due to erosion (t/yr)
1.1.1	1 084 860	32 341	32 341	6468	-25 873
10	137 082	176 168	176 168	35 234	-140 935
11	477 789	68 563	68 563	13 713	-54 851
12	39 944	1127	1127	225	-901
2.1.1	64 435	3656	3656	731	-2925
3.1.1	76 839	8152	8152	1630	-6522
4.1.1	1 348 242	98 735	98 735	19 747	-78 988
4.2.1	358 195	11 623	11 623	2325	-9298
5.1.1	323 932	35 973	35 973	7195	-28 778
5.2.1	836 692	8965	8965	1793	-7172
5.3.1	183 841	2465	2465	493	-1972
5.4.1	169 148	4233	4233	847	-3386
5.4.2	392 540	4155	4155	831	-3324
5.4.3	6378	26	26	5	-21
6.1.1	169 292	39 489	39 489	7898	-31 591
6.2.1	244 784	8148	8148	1630	-6519
6.3.1	172 255	4983	4983	997	-3986
6.3.2	91 823	23 636	23 636	4727	-18 909
6.3.3	241 674	14 915	14 915	2983	-11 932
6.3.4	26 572	1024	1024	205	-819
6.4.1	643 255	8507	8507	1701	-6806
6.4.2	406 442	3237	3237	647	-2589
6.4.3	81 638	1431	1431	286	-1145
6.4.4	29 397	241	241	48	-193
7.1.1	22 086	3498	3498	700	-2799
7.1.2	271 767	129 020	129 020	25 804	-103 216
7.1.3	72 786	8133	8133	1627	-6507
7.2.1	520 643	9304	9304	1861	-7443
7.2.2	73 406	2896	2896	579	-2317
7.2.3	136 893	3863	3863	773	-3091
7.2.4	77 158	1493	1493	299	-1195
7.2.5	50 549	693	693	139	-554
7.3.1	240 821	5310	5310	1062	-4248
7.3.2	18 116	1374	1374	275	-1099
8.1.1	2 614 304	505 585	505 585	101 117	-404 468
8.1.2	1 625 992	1 403 840	1 403 840	280 768	-1 123 072
8.1.3	1 303 021	58 775	58 775	11 755	-47 020
8.1.4	132 913	21 767	21 767	4353	-17 414
8.1.5	26 101	1764	1764	353	-1411
9	322 206	144 296	144 296	28 859	-115 437
Total	15 115 811	2 863 405	2 863 405	572 681	-2 290 724

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CHAPTER SIX IMPLICATIONS OF POST-2012 LULUCF ACCOUNTING OPTIONS

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Summary

This chapter evaluates the contribution of the LULUCF sector to New Zealand's net position under the major options being proposed for post-2012 accounting. Six different accounting options have been considered:

- (i) A status quo (Gross-Net) approach to LULUCF accounting.
- (ii) An All-Lands Gross-Net Accounting approach.
- (iii) An All-Lands Gross-Net Accounting approach with an additional cap on credits/debits from forest management.
- (iv) An all-lands net-net accounting approach referenced to 1990.
- (v) A forward-looking baseline approach.
- (vi) The Average Carbon Stocks (ACS) approach.

Analysis of the LULUCF Sector of New Zealand's budget over the First Commitment Period clearly indicates that accountable net emissions are dominated by carbon uptake by exotic forests and carbon losses from deforestation. Carbon stock changes associated with cropping, pastoral agriculture, horticulture and forest management presently make quantitatively small contributions to New Zealand's overall reported position, either because net stock changes per units area are small or because the areas involved are small. The possible exception to this is pastoral agriculture: present results suggest gains in some landscape/ production systems and losses in others, but uncertainty remains high, and small overall net gains or losses are possible. As pastoral agriculture covers such a large area of New Zealand, even relatively small changes per unit area could add to a significant amount for the country as a whole.

For the Second, Third and further Commitment Periods, New Zealand's LULUCF emissions are expected to continue to be dominated by exotic plantations, deforestation and, provided that sufficient policy incentives are provided, the reestablishment of indigenous forests especially on marginal pastoral land. Whether deforestation rates will remain as high as in the recent past will depend on future trends in economic drivers, such as wood and dairy prices, and the extent to which disincentives, such as through the ETS, are maintained and enforced over future Commitment Periods. It is also again necessary to be cautious with respect to possible changes in soil carbon under pastoral agriculture, as any possible changes are yet to be adequately quantified.

Across the range of plausible mitigation options and accounting schemes, the Sector as a whole could potentially contribute large credits or debits. Plausible ranges were calculated as -120 to +125 Mt CO₂ over the Second Commitment Period, and between -135 and 120 Mt CO₂ over the Third Commitment Period. This range is about equally due to possibilities for the success of mitigation options, such as the large potential for forest establishment and preventing deforestation, as to the range of possible accounting options.

As scenarios, it was assessed that future establishment rates of exotic forests might range between 0 and 40 000 ha per year, establishment of indigenous forests could potentially range

between 0 and 100 000 ha per year, and deforestation rates could vary between 0 and 10 000 ha per year. Together, these options could have a combined mitigation potential of 90 Mt CO₂ over the Second Commitment Period and 140 000 Mt CO₂ over the Third Commitment Period.

In terms of possible accounting options, continuation of the status quo (Gross-Net accounting), 'All-Land Gross-Net Accounting with a Cap on Forest Management Emissions', and application of the 'Forward-Looking Baseline' approach would lead to similar outcomes for the LULUCF Sector over the Second Commitment Period, and would be likely to generate credits similar to those anticipated for the First Commitment Period. These credits would diminish for the Third Commitment Period as the existing post-1990 estate reaches maturity and no longer generates further credits. On-going credits could only be maintained through substantial new plantings.

New Zealand's net position would, however, be much worse under the 'All-Land Gross Net Account' (without a cap) because pre-1990 exotic forests would need to be included and these forests are anticipated to constitute a significant source over the Second and Third Commitment Periods. The worst possible outcomes for New Zealand would occur under application of 'All-Lands Net-Net Accounting because New Zealand's forests were a large sink in 1990 (by 95 Mt CO₂ per Commitment Period). If that uptake had to be included in the baseline it would worsen New Zealand's net position by those 95 Mt CO₂.

Application of the Average Carbon Stocks approach would lead to lower credits for the Second Commitment Period than the most beneficial options. That differences between the Average Carbon Stocks approach and the other options becomes small by the Third Commitment Period as the post-1990 estate matures.

Overall, the range of possibilities due to uncertainties in the success of future mitigation policies, and about possible accounting options, far outweighs the scientific uncertainty about specific processes or extent of specific activities (with the possible exception of carbon-stock changes under pastoral agriculture). The outstanding priorities are therefore to:

- develop better scenario assessment tools to quantify the consequence of different accounting options and assess their implications for New Zealand's net position;
- more completely quantify changes in soil carbon stocks under pastoral agricultural management regimes and soil types (including for forage cropping under low/no-till options);
- better determine the effects of cropping on soil carbon stocks, as cropland has become a key category for New Zealand.

1. Introduction

The Kyoto Protocol set a number of specific rules for the accounting of biospheric carbon stocks. These rules were developed after many late-night negotiations, under intense time pressure and without thorough knowledge and understanding of some of the key issues and their quantitative implications. Many of these issues, problems and possible accounting anomalies were subsequently identified and highlighted by a Special IPCC Report on Land Use, Land-Use Change and Forestry (IPCC 2000).

There have been many subsequent discussions about possible accounting options to replace the current rules of the Kyoto Protocol for future Commitment Periods (Schlamadinger et al. 2007a; Höhne et al. 2007). A number of researchers have made quite specific proposals that could replace the current accounting rules (e.g., Cowie et al. 2007; Schlamadinger et al. 2007b; Böttcher et al. 2008), but none have so far been adopted. A key challenge in developing such rules is to identify means by which anthropogenic and non-anthropogenic components can be distinguished (Canadell et al. 2007).

Despite this significant body of work, the international negotiating community is yet to settle on any preferred set of rules to be used for a Second and subsequent Commitment Period for the Kyoto Protocol. In fact, negotiations appear so slow there seems to be little movement towards a preferred position, or any move to dismiss any of the possible options.

For New Zealand to continue to participate actively and effectively in these international negotiations, it is necessary to have a good appreciation of the effect of different accounting options on New Zealand's net greenhouse gas position so that the negotiating team can take an informed position on any possible proposal. Below, we briefly discuss and quantify the implications of six possible accounting options for the LULUCF sector on New Zealand's net position. The analysis does not consider fossil fuel-based emissions or emissions of non-CO₂ greenhouse gases. It must also be emphasised that this is not a complete list of possible accounting options, but we have covered the main options presently under consideration. Considering even these accounting approaches reveals a large range of possible outcomes, and highlights the importance of being well-informed on the critical implications of different accounting options.

2. Scope of Study

In this chapter we have developed an integrated analysis of the effect post-2012 accounting options are likely to have on New Zealand's net position under the Kyoto Protocol, for the expected range of future LULUCF activities and mitigation options. The work is based on the datasets, forecasts, mitigation options, and measures of uncertainty developed in Chapters 3 to 5. The following accounting options are analysed:

- (i) A status quo (i.e. Article 3.3 only) approach to LULUCF accounting.
- (ii) Retention of Article 3.3 and adoption of Article 3.4 accounting of emissions and removals under grassland management, cropland management, forest management, and revegetation.

- (iii) The same as 2, but with an additional cap on credits/debits from forest management.
- (iv) An all-lands net-net accounting approach referenced to 1990.
- (v) A forward-looking baseline approach where net emissions under a business-as-usual scenario are calculated for a Commitment Period and actual net emissions with additional measures are then compared against the business-as-usual baseline.
- (vi) The Average Carbon Stocks (ACS) approach where accounting is based on the assessed time-averaged carbon stocks associated with specified land-use/land-management combinations.

The analysis has been segregated by the LULUCF sub-sectors that are considered to be the most quantitatively important contributors to current and likely future emissions or uptakes. This is to provide further assistance with prioritisation of future work to address knowledge gaps and uncertainty.

To begin the analysis, we summarise the key components that contribute to New Zealand's position during the First Commitment Period, CP1 (Table 1). While these numbers are our current best estimate, they include elements of uncertainty due both to elements of the science as discussed in detail in this report, and to activity data during CP1 itself. This is particularly important for deforestation losses. As the bulk of deforestation emissions are assumed to be released in the year of deforestation (according to the default accounting methodology), the actual deforestation rates over the remaining years of CP1 will be the key driver of those emissions, and emissions cannot be reported with confidence until actual deforestation rates are reported after the end of the Commitment Period. The numbers in Table 1 are thus close, but not identical, to the numbers reported in New Zealand's latest Net Positions Paper (MfE 2008).

Table 1 Land-based reportable CO₂ uptakes and emissions during CP1, separated into biomass C and soil organic carbon (SOC). Numbers are in Mt CO₂ per 5-year CP.

	Biomass C	SOC	Total
Exotic forestation since 1990	89	-7.5 ¹	81.5
Deforestation (mainly exotic forests)	-17	1 ²	-16
Indigenous forestation	0	0	0
Established indigenous forests	≈0	0	0
Forest management	≈0	≈0	0
Pasture	0	≈0	0
Cropping	0	-0.1	-0.1
Horticulture	≈0.3 ³	-0.3	≈0
Total	72.3	-6.9	65.4

¹Based on an assumed 12 t C ha⁻¹ loss and using the detailed analysis in Table 7, Chapter 3, to convert it to a CO₂ loss over the commitment period.

²Assumes 12 t C ha⁻¹ gain over 20 years. This estimate is very uncertain.

³We are not aware of any national estimate of biomass C stocks under horticulture. The number will vary with crop type, being reasonably high for tree crops, like apples, but much lower for grape vines.

During CP1, accountable carbon uptakes and emissions are dominated by forestation²¹ with exotic forests, mainly by *Pinus radiata* plantations, established mainly in the early 1990s. The trend in more recent years has been much reduced planting rates and some conversion of forest back to pastoral land, mainly driven by the dairy boom over recent years. Deforestation has therefore become numerically the second-largest item in the national carbon budget.

Future deforestation rates for even the remainder of the First Commitment Period remain highly uncertain. New Zealand's latest Net Positions Paper (MfE 2008) reports that 19 000 ha had been deforested in 2007 and assumes that a further 12 000 ha would be deforested over the First Commitment Period. While the economic drivers are likely to favour further conversion from forestry to dairying over the foreseeable future, deforestation could be largely curtailed by effective penalties through the ETS. Whether high deforestation rates will thus eventuate will depend largely on future policies with respect to maintaining, strengthening or softening the ETS.

Other land uses are thought to have undergone only small net carbon-stock changes, and whatever small changes these were, they need not be accounted for under Article 3.3. Article 3.4 was not compulsory for CP1, and New Zealand chose to not include these optional components.

For the Second and subsequent Commitment Periods, it is anticipated that New Zealand's carbon budget will continue to be dominated by carbon changes in exotic forests and deforestation. In addition, initiatives are being introduced to encourage the reestablishment of forests, both exotic and indigenous. Future initiatives may increasingly target forest establishment on poorly productive and erosion-prone marginal pastoral hill country. The following discussion of different accounting options will concentrate primarily on deforestation and forestation with exotic and indigenous forests. Carbon-stock changes on land under other land uses may well have to be accounted in future, but since in New Zealand's case the associated carbon-stock changes are likely to be small, the effect of different rules on the accounting under those different land uses is unlikely to materially affect New Zealand's overall position.

A possible exception involves C-stock changes under pastoral land use, chiefly because the areas involved are so large. At this stage, the best available information suggests the sector as a whole is carbon neutral even though there appear to be compensating trends in different sub-sectors (see Chapter 5.5). If future work can substantiate whether there are indeed consistent trends in soil carbon associated with specific land forms or production systems, it might present significant mitigation opportunities or, conversely, large liabilities if land management options cannot be modified to avoid carbon-stock losses. At this stage, however, it is not certain that C-stock changes are occurring widely, and if they do occur, what they are caused by. It is therefore not possible to assess meaningfully the effect of different accounting options on any trends that are still so scientifically uncertain, particularly as the current balance of evidence suggests that the Sector is carbon neutral overall.

The other major flux is that from exotic plantations that were established before 1990. Under the wording of Article 3.3, carbon stock changes in these forests do not need to be accounted

²¹ Forestation is used here as a generic term to encompass both afforestation, which is applicable on land that had never been forested, and reforestation, which refers to land which historically had a forest cover. The distinction between afforestation and reforestation carries no practical implication for accounting purposes.

for CP1, but under some accounting rules, that situation might change in future Commitment Periods.

Because of the 1990 cut-off year, and because there was a high establishment rate around 1990, both the pre-1990 and post-1989 exotic estates have skewed age-class distributions, with consequent well-defined periods of strong carbon uptake and release over the coming decades (see Figure 10, 11; Chapter 3). Both estates are currently considered to be C sinks. However, the pre-1990 estate is expected to turn into a net source by about 2010 and the post-1990 estate will become a net source by about 2020. As a consequence, it may become very important whether the pre-1990 estate should be included in future accounting. The treatment of wood products is also important in this context. This issue is not further dealt with below, but an extended discussion of the consequences of different accounting options is provided in Chapter 3.

The following discussion thus primarily focuses on the effect of different accounting options on New Zealand's overall net position. However, while it is not explicitly addressed in the following discussion, it is also important to remain cognisant of the effect of different accounting options in encouraging the adoption of land-use / land-management options with beneficial environmental outcomes even if they may be numerically small (Schlamadinger et al. 2007a). For example, given the voluntary nature of Article 3.4, the accounting rules under the Kyoto Protocol currently provide no incentives for increasing carbon stocks under pastures or forests other than those that have undergone a land-use change since 1990. It is important that future accounting rules are tailored to encourage all management options that could be used to cost-effectively increase carbon stocks.

3. The Key Accounting Options

In the following sections, we briefly discuss the key elements of the most likely accounting options to be adopted post-2012 before assessing the effect on New Zealand's net position under each of these options.

3.1 Gross-Net Accounting (Status Quo)

Forest-based activities have been included in carbon accounting through Articles 3.3 and 3.4 of the Kyoto Protocol. Article 3.7 is relevant for some countries, notably Australia, but not for New Zealand. This set of rules has been criticised for internal inconsistency, the creation of emissions loop-holes, and for the omission of a range of potentially important activities (Schlamadinger et al. 2007a, b; Cowie et al. 2007). Despite these various short-comings, it is possible that the international negotiating community will not be able to agree on any replacement accounting scheme and the existing set of rules will continue to be used as the default position into the future.

In essence, this would mean that zero baseline net emissions will continue to be applied ("gross" accounting), and that only carbon fluxes in stands where new forests have been established since 1990, or deforested stands that were forests in 1990 will be counted. The artificial distinction between Kyoto forests and non-Kyoto forests would thus continue, and would continue to provide no incentive for carbon management in pre-1990 forests.

3.2 All-Lands Gross-Net Accounting

The Kyoto Protocol stipulates that Article 3.4 activities are voluntary for CP1, but are expected to be binding for CP2 and subsequent Commitment Periods, although the Protocol was not specific as to the type of activities to be included under the broad category of 'human-induced activities'. Through subsequent decisions by a number of conferences of the Parties (COPs), especially the Marrakech Accord, these activities have been further defined, such as forest management, cropland management and grazing-land management. However, it is yet to be confirmed whether these activities will, indeed, become binding for future Commitment Period, or whether further negotiations, or clarification of the specific meaning of the text of the Kyoto Protocol, will lead to these activities to continue to be treated as voluntary, or whether some other approach may emerge.

The meaning of forest management appears to have shifted towards an understanding that once forest management is included, all carbon-stock changes in managed forests are deemed to be included as human-induced activities. In other words, all carbon-stock changes in all managed forests would be counted, irrespective of their causes because it has so far proven elusive to try and find practical means of factoring out a human-induced component to any carbon-stock changes. For New Zealand, this would principally mean inclusion of pre-1990 exotic forests. Indigenous forests would also be included, but since the best current assessment (see Chapter 4) is that these forests are currently carbon neutral, their inclusion would not alter New Zealand's net position.

3.3 All-Lands Gross-Net Accounting with a Cap on Net Emissions

A major problem with inclusion of all forest activities is that the forests of some countries can undergo major fluctuations without any current or recent management cause, but simply due to their historic age-class distribution (e.g., Böttcher et al. 2008), which is most pronounced in the case of Canada (Kurz & Apps 1999; Kirschbaum & Cowie 2004). Increasing carbon stocks can also be due to regrowth of forests after reforestation in previous decades, which is very pronounced in the USA (Kirschbaum & Cowie 2004).

Consequently, as part of the Marrakech Accord, caps were introduced on allowable credits (and presumably debits) providing a floor level to prevent windfall gains for countries with large increases in the carbon stocks in their forests. It is understood that the initial intent of the caps had been to provide a limit on allowable credits, but the wording has been such that debits should be covered as well. It would thus provide a protection for countries where forests might have undergone large carbon losses for natural (e.g. wildfire, insect damage) or human-induced (e.g. regular harvesting) reasons. The cap does not cover carbon losses due to land-use change as land conversion would trigger the application of Article 3.3 which is not covered by a cap.

The agreed cap for New Zealand is 0.2 Mt C yr^{-1} or about $4 \text{ Mt CO}_2 \text{ CP}^{-1}$. The existing agreement covers only the First Commitment Period, and the same caps might be used for subsequent Commitment Periods, or their numeric values could be altered through future negotiations, or the wording might be tightened to define the caps to apply only to removals. In analysing this basic accounting option, we have assumed that the existing cap would also be applied in future Commitment Periods, but we recognise that it could be changed in a number of ways which could also significantly alter the net position for New Zealand.

We also assumed that such a cap would be applied only to those forest activities not covered under Article 3.3. Hence, forests planted since 1990 could be accounted without any cap, but carbon gains and losses from pre-1990 forests, including much of the exotic forest estate and practically all indigenous forests, would be subject to the cap.

3.4 All-Lands Net–Net Accounting

All-lands net–net accounting is very similar to ‘All-Lands Gross-Net Accounting’ except that net emissions in a base-line year are also accounted, and credits/debits are assessed as the difference between net emissions in the Commitment Period compared with emissions in the baseline year. The base year could, in principle, be 1990, or any other year, or it could be a rolling base year that increments by 5 years every Commitment Period. Such accounting is discussed here with 1990 as the base year, but it must be recognised that the options of using other base years is also being discussed internationally, and that this could significantly alter the net outcome for New Zealand.

Net–net accounting has been seen as a simple, but very crude way of factoring out age-class legacy effects and identifying a human-induced emissions component (Schlamadinger et al. 2007b; Cowie et al. 2007; Canadell et al. 2007). There are, however, numerous problems with using net–net accounting for this purpose (Cowie et al. 2007), which were the principle reasons why net–net accounting was not originally adopted for the Kyoto Protocol (even though it is used for fossil-fuel emissions).

It is hoped net–net accounting will not be reconsidered. However, it is attractive in its conceptual simplicity, which might an impetus for its reconsideration in future.

3.5 Forward-Looking Baseline Approach

A forward-looking baselines method has been presented at UNFCCC meetings (e.g., https://unfccc.int/files/meetings/ad_hoc_working_groups/kp/application/pdf/accra_pres_lulucf_canada.pdf) and formally been proposed in Canada’s submission to the UNFCCC in August 2008 (http://unfccc.int/files/kyoto_protocol/application/pdf/canada.pdf), with further elaboration provided in a further UNFCCC submission in November 2008 (http://unfccc.int/files/kyoto_protocol/application/pdf/canadalulucfkp271108.pdf). The method as described in Canada’s submission appears essentially to be a simplified version of the ‘Net Accounting With Negotiated Baselines’ approach described by Cowie et al. (2007). A partial description has also been provided by Böttcher et al. (2008). As we understand the approach, it is intended to cover only forest management and would essentially consist of the following steps:

- (i) Before a Commitment Period, it would be necessary to estimate a ‘Business-As-Usual’ net emissions baseline of the emissions that would be expected over a Commitment Period based on the age-class of the existing forestry estate, typical harvest ages, and other expected silvicultural practices.
- (ii) After the end of the Commitment Period, the baseline could be adjusted for any unanticipated natural events beyond the control of land managers, such as losses due to wildfires, volcanic eruptions, insect or disease outbreaks or anything else deemed outside the control of land managers.

- (iii) Actual net emissions would then be compared against the adjusted baseline, and credits or debits would be assigned based on that difference. The adjusted baseline would include known legacy effects, such as the area and age-class distribution of the existing estate, and any additional natural factors that can cause carbon gains or losses.

While the proposal seems to work fairly well in eliminating problems caused by the legacy effect of uneven age structures, it is not clear how the proposal might work for land-use changes. It is therefore only applied to forest management, and land-use change is accounted in the same way as under the current status quo option.

3.6 Average Carbon Stocks Approach

The Average Carbon Stocks (ACS) approach for biospheric carbon accounting is based on estimating time-averaged carbon stocks under different land-use and management systems and then assigning debits or credits based on the differences in average carbon stocks that result from shifts in land use or management (Kirschbaum et al. 2001; Kirschbaum & Cowie 2004; Cowie et al. 2007). The approach is essentially the same as the approach adopted by New Zealand for soil-carbons stocks, but extends further to include plant biomass carbon stocks as well. Key advantages of this approach are its simplicity and low compliance costs. Under the ACS approach, a nation's land area would be sub-divided into defined biosphere domains. Each domain would be categorised by its land use and would include a category for unmanaged land, and its soil type or climatic zone to the extent that this would be warranted by differences in carbon-storage potential across a land-use type. For each biosphere domain, average carbon stocks would be determined for the defined land use under its typical management practices. For example, for a production forest, this would be the average of carbon stocks over typical multiple rotations under standard silvicultural management.

Changes in average carbon stocks due to land-use or management changes lead to equivalent credits or debits for the responsible land holder. These would primarily result from changes in the land area in respective land-use categories with different average carbon stocks, or, to a lesser extent, from changes in the average carbon stocks within defined land-use categories.

To calculate the amounts of carbon credited or debited over a Commitment Period, the calculated changes in average carbon stocks need to be apportioned over the time it takes for biospheric carbon stocks to reflect the new land use, as is presently done for adjustments in soil-carbon stocks. Hence, creditable carbon stocks would approximate realised carbon-stock changes until the average carbon stocks under a new land use are reached, and then remain constant irrespective of actual short-term increases and decreases that might occur (e.g., due to harvesting), provided the long-term average carbon stocks remain the same.

4. Comparison of the Key Options

Here we compare New Zealand's net position under five key accounting options. The generated numbers are indicative only but they are intended to be sufficiently accurate to allow a meaningful comparison between the different options and potential mitigation scenarios.

4.1 Underlying Assumptions

Derivation of the data in Tables 2 and 3 was based on the following key considerations:

- (i) The analysis includes exotic forests and deforestation because they are currently the dominant contributors to New Zealand's overall net position, and the additional forestation of indigenous forest on marginal land that can potentially become a significant contributor if the identified forestation opportunities are utilised to their fullest. The analysis excludes carbon-stock changes associated with cropping, horticulture and pastoral agriculture because contributions from these sectors to New Zealand's total net emissions are very small either because the areas are small or because there are no identified changes in carbon stocks (Table 1).
- (ii) The underlying data for exotic forests have been presented in Chapter 3 of this report. Two numbers are presented as scenarios to illustrate the full extreme range of potential establishment rates, corresponding to planting rates of 0 and 40 000 ha yr⁻¹. Chapter 3 lists wider and more detailed scenarios, with a calculated sequestration potential of 25 Mt CO₂ CP⁻¹ for CP2 and 60 Mt CO₂ CP⁻¹ for CP3. All these calculations are based on the best currently available models and inventory data, and predicated on the continuance of current forestry practices. With further refinement of models or their underlying data, or if forest operations were to change in response to incentives under the ETS, for example, calculated numbers for New Zealand's net position would obviously change.
- (iii) For calculating the credit from exotic plantations under the ACS, we assumed that plantations would accrue credits for 15 years before reaching the average stocks of exotic forests, after which no further credits or debits would accrue. We used the annual establishment rates of post 1989-forests (see Chapter 3) to calculate the proportion of the estate that was still within the initial 15-year window at respective future dates.
- (iv) For replanting with indigenous forests, the options considered include no replanting and replanting to the full identified extent of 4.6 Mha. It was assumed here that as an upper extreme, about one tenth of this area could be replanted by the middle of CP2 for a sequestration potential of 15 Mt CO₂ CP⁻¹ and a further tenth by the middle of CP3 for a combined sequestration potential of 30 Mt CO₂ CP⁻¹. To achieve that potential would require establishment rates of nearly 100 000 ha yr⁻¹, which is considered to be the upper range of forestation rates that might be achievable with sufficient policy incentives.
- (v) For deforestation, we used two assumptions: a deforestation rate of 10 000 ha per year, or complete cessation of deforestation. We took a carbon loss of 1000 t CO₂ ha⁻¹ for the carbon loss from a mature and fully stocked stand. Future deforestation rates are very difficult to predict as they depend on future trends in commodity prices balanced by political pressure and regulations to discourage deforestation activities.
- (vi) For the 'All-Lands Gross-Net With A Cap' option, it was assumed that the carbon stock changes arising from forests established before 1990 (either indigenous or exotic), would be capped at the cap agreed to at COP 7. It is also assumed that the cap applies to both gains and losses. For New Zealand, that cap is 0.2 Mt C yr⁻¹ (about 4 Mt CO₂ CP⁻¹) so that application of the cap renders inclusion of pre-1990 forests nearly

irrelevant. If the cap were applied only to carbon gains, the outcome would revert to be identical to that without a cap as the pre-1990 estate is expected to become a large carbon source over the Second and Third Commitment Periods.

4.2 Key Outcomes

The key issues that arise by looking at New Zealand's future position are:

- (i) During CP1, New Zealand expects to gain credits of about 65 Mt CO₂ CP⁻¹ from carbon-stock changes associated with land-based activities (Table 1), with credits of 81.5 Mt CO₂ CP⁻¹ from exotic plantations established since 1990. Without substantial new plantings, this credit will be maintained for CP2 (Table 2) but will erode substantially for CP3 (Table 3). This trend is simply related to the aging of the estate and the trees' maturing to their typical harvestable age. Under the current accounting methodology, and without considering any new plantings, the credit from post-1989 exotic forests could be about 85 Mt CO₂ CP⁻¹ for CP2, but then fall sharply to 30 Mt CO₂ CP⁻¹ for CP3.
- (ii) Substantial credits can be maintained into the future if planting rates of both exotic and indigenous forests can be brought to their assessed potentials. The current economic drivers make commercial forestry relatively less economic than dairying, in particular. If those economic drivers continue on their current trends, it might be difficult to increase the planting rate of exotic forests, and, indeed, the recent trend has been for a reduction of the commercial forestry estate. There is, however, considerable potential to increase the establishment rate of indigenous forests as there is a large area of economically marginal land that could be reforested for little lost economic opportunities and substantial environmental co-benefits (see Chapter 4). To achieve large sequestration rates, it would be necessary to reforest substantial areas of land as sequestration rates per unit of land are relatively small and much less than for exotic forests.
- (iii) Future deforestation rates are also very important, mainly because each conversion can lead to an assumed large immediate release of a very large amount of carbon per unit of land. The ETS Policy paper has estimated deforestation rates between 2400 and 7400 ha yr⁻¹ through to the end of CP1. Deforestation rates further into the future can potentially cover a wider range between complete cessation if it is aggressively discouraged through appropriate policy intervention or potentially increase to 10 000 ha yr⁻¹ if economic drivers continue to favour the expansion of pastoral agriculture and if such expansion is not discouraged by policy intervention. These are not very large areas, but have disproportionately large carbon implications. Carbon implications are not quite as dramatic if the fate of harvested wood products were included in the accounting, and the implications of a range of potential approaches are discussed in Chapter 3.

Table 2. New Zealand's net position during the Second Commitment Period (CP2) under five key accounting options and broken down by exotic forests, re-planting of indigenous forests and deforestation. This is shown for optimistic and pessimistic planting rates. All numbers are in Mt CO₂ per Commitment Period. Planting and conversion rates are in ha yr⁻¹. 'BL' refers to the baseline amounts that have to be applied under respective accounting options. Lower and upper totals add the most pessimistic (lowest forestation and highest deforestation rates) and most optimistic values, respectively, from the individual columns after subtracting the respective baseline values.

	Exotic forest			Indigenous forests			Deforestation			Total		
		Planting rate			Planting rate			Conversion rate				
	BL	0	40 000	BL	0	100 000	BL	10 000	0	Lower	Mean	Upper
Post-1990 gross-net	0	85	110	0	0	15	0	-50	0	35	80	125
All forest gross-net	0	25	50	0	0	15	0	-50	0	-25	20	65
All forest gross-net w. cap	0	81	106	0	0	15	0	-50	0	31	76	121
All lands net-net	95	25	50	0	0	15	0	-50	0	-120	-75	-30
Forward looking baseline	-75	10	35	0	0	15	0	-50	0	35	80	125
ACS	0	16	41	0	0	15	0	-25	0	-9	23.5	56

Table 3 New Zealand's net position during the Third Commitment Period (CP3). Details as for Table 2.

	Exotic forest			Indigenous forests			Deforestation			Total		
		Planting rate			Planting rate			Conversion rate				
	BL	0	40 000	BL	0	100 000	BL	10 000	0	Lower	Mean	Upper
Post-1990 gross-net	0	30	90	0	0	30	0	-50	0	-20	50	120
All forest gross-net	0	10	70	0	0	30	0	-50	0	-40	30	100
All forest gross-net w. cap	0	26	86	0	0	30	0	-50	0	-24	46	116
All lands net-net	95	10	70	0	0	30	0	-50	0	-135	-65	5
Forward looking baseline	-25	5	65	0	0	30	0	-50	0	-20	50	120
ACS	0	1	61	0	0	30	0	-25	0	-24	33.5	91

- (iv) At its most optimistic, with substantial expansion of the forest estate and a complete cessation of deforestation, New Zealand's net position could significantly improve into the future, with total net credits increasing from 65 Mt CO₂ CP⁻¹ for CP1 to 125 Mt CO₂ CP⁻¹ for CP2, and to 120 Mt CO₂ CP⁻¹ for CP3. This would be an extremely optimistic scenario. A more realistic scenario (given by our mean estimates) would be an approximate maintenance of New Zealand's current position with a slight increase of credits to 80 Mt CO₂ CP⁻¹ for CP2, falling slightly to 50 Mt CO₂ CP⁻¹ by CP3. It must be emphasised that even this mean estimate can only be achieved with substantial policy initiatives that would require, for example, planting of 20 000 ha yr⁻¹ of exotic forests, establishment of 50 000 ha yr⁻¹ of indigenous forests, and keeping deforestation rates to 5 000 ha yr⁻¹ (or some other compensating combination of measures).
- (v) New Zealand's net position is only marginally affected by soil C changes under cropping, horticulture or pastoral agriculture, or with silvicultural options in forestry. Soil C changes with land-use change do partly negate the opposite biomass C-stock changes. While biomass C-stock changes are much larger, soil C stock changes are nonetheless large and important enough to be considered as well. Changes in carbon stocks under pastoral agriculture are not currently considered because the sector as a whole is currently assessed to be carbon neutral (with possible C-stock losses in flat dairy land and gains in sheep-grazed hill country). However, future work is warranted to ascertain whether these trends are indeed compensating, whether one dominates, or whether a shift in activities might lead to a trend towards an overall gain or loss. If any divergent trends can be further substantiated and better understood, they might provide future mitigation opportunities or, conversely, might need to be reported as carbon losses associated with particular management types. Because of the large areas involved (13 Mha), even small changes could have an important influence on the national budget. As no current overall trend has been identified, it is not currently included in assessing New Zealand's future position.
- (vi) In terms of carbon accounting options, the option that clearly stands out in terms of strongly disadvantaging New Zealand would be all-lands net-net accounting. It would greatly change New Zealand's net position for two reasons: first, it would reduce New Zealand's net position by about 95 Mt CO₂ CP⁻¹ because that was (five times) the estimated carbon uptake by New Zealand's forest sector in 1990; second, as many of New Zealand's pre-1990 plantations are reaching their harvestable age by CP2, this estate is likely to contribute a large carbon loss if it needs to be included. The loss is particularly large over CP2 (at about 75 Mt CO₂) before decreasing to about 25 Mt CO₂ over CP3. The estate would then either become a sink again over even later Commitment Periods or approach carbon neutrality if the age-class distribution was to eventually even out.
- (vii) The most advantageous position in the short term would be continuance of the status quo in which only post-1989 plantations would be counted, and without applying a baseline. Post-1989 exotic plantations are expected to contribute 85 Mt CO₂ over CP2 and 30 Mt CO₂ over CP3, even without planting any new stands. Benefits from that estate would then disappear, however, as post-1989 stands would also reach their harvestable age and the estate would undergo a period of carbon loss.
- (viii) An extension of accounting from the post-1989 estate to all forests would reduce New Zealand's net position because the anticipated losses from the pre-1989 would need to be included. Those losses would be minimised if New Zealand's cap on forest

management (0.2 Mt C yr^{-1}) could be applied, if that cap was interpreted to cover both gains and losses from established forests, and if the post-1990 estate was not covered by the cap.

- (ix) Application of a forward-looking baseline would basically negate all C-stock changes in New Zealand's forests other than those arising from new plantings (which is what the approach is intended to achieve). This would prevent large liabilities developing from C-stock changes in the pre-1990 estate (unlike the situation under net-net accounting). As presented by Canada's submission, the forward-looking baseline would be applied only to pre-1990 forests, but post-1990 forests would continue to be covered under Article 3.3.
- (x) The overall position would be reasonably favourable under the Average Carbon Stocks (ACS) approach because some benefit of already established plantations would still continue, especially for CP2 but to a lesser extent for CP3. Beyond CP3, credits would only accrue to the extent that new forests were established. An important safe-guard under ACS would be that on-going carbon increases and decreases within the same land-use classes would create neither gains nor liabilities.
- (xi) It is expected that all accounting options would treat the forestation of indigenous forest in essentially the same way and would give credits that equate with the associated increase in carbon stocks. The only proviso would be that under some definitions of 'direct-human induced' (or equivalent wording), it might be problematic to get credits for natural re-establishment of forests through natural seeding after fencing-off stock or other minimal-intervention strategies that would rely largely on natural processes, with the principal human-induced effect being the removal of grazing stock. Provided this will not be a problem, the options would not differ with respect to crediting natural forest re-establishment
- (xii) Similarly, extension of exotic forests on previous pasture land would be treated identically by most of the different options. Apart from the ACS, all other options would estimate the carbon stock increment as it accrues and assign credits accordingly. Under the ACS, the same credits would accrue but only up to the time when average carbon stocks are reached, and accountable stocks would remain constant from that point on. For plantings from 2008 onwards this would cause no difference between the ACS and the other schemes. Credits under the ACS, however, would start to saturate sooner (from CP4 onwards) than for the other schemes. On the other hand, there would be no need to account for C losses when stands are harvested (provided that they are re-planted after harvesting).
- (xiii) For deforestation, there are again few differences between the accounting options, provided the forward-looking baseline approach is used only for forest management and land-use changes are accounted with an approach essentially equivalent to that used under the current status quo approach. Under the ACS approach, however, deforestation (of managed forests) would lead to debits of only about half the magnitude of those under the other schemes because it would be calculated on the basis of average carbon stocks rather than on the basis of the stocks at the end of a rotation when they would be at their maximum. These lesser losses would, of course, be balanced by lesser gains in any earlier accrual period.

4.3 Implications of Post-2012 Accounting Options

Depending on the mix of policy options that may lead to greater or lesser forestation and deforestation rates and the range of possible accounting options, the LULUCF sector may contribute between -115 and $+125$ Mt CO₂ CP⁻¹ for CP2, and between -136 and 120 Mt CO₂ CP⁻¹ for CP3. This is a huge range and is almost equally affected by the success of implementing mitigation options as it is by the range of possible accounting options.

Among the accounting options, differences principally relate to the treatment of exotic forests because new land-use conversions between forest and pasture are treated in much the same way by virtually all accounting options. The one exception are carbon losses due to deforestation, where the ACS would account C losses from the average carbon stocks held by a forest compared with the other accounting options that (are assumed to) account for carbon losses from the carbon stocks that are present just before land conversion. This means that under the ACS, accountable losses would be only about half those under the other accounting options. These lesser losses under the ACS are partly balanced by lesser gains during the establishment phase of new forests.

For the treatment of exotic forests, the two principal differences between accounting options are whether a base line uptake has to be included, and whether carbon-stock changes in pre-1990 stands need to be included. The one accounting option that clearly stands out is net-net accounting. New Zealand's exotic forests were a strong sink in 1990, and if that needs to be included as a baseline uptake in the accounting, it clearly makes a huge difference (by 95 Mt CO₂ CP⁻¹).

It also makes a big difference whether the pre-1990 estate needs to be included or not (compare the 'Post-1990 forests gross-net' and 'All forests gross-net' options). The difference is particularly large for CP2 because this is the time when many pre-1990 stands are expected to reach their harvestable age and when carbon stocks are consequently expected to fall drastically. These dramatic falls would be softened if the fate of wood products were explicitly included. The inclusion of pre-1990 forests also becomes fairly insignificant if a cap on stock changes in managed forests (of 0.2 Mt C yr⁻¹ for New Zealand as agreed to as part of the Marrakech Accord). This assumes the cap is applied only to pre-1990 forests (Article 3.4 activities), the cap applies equally to gains and losses, and its numeric value does not change through further negotiations. The validity of these assumptions will depend on the outcome of future international negotiations.

A characteristic of New Zealand's exotic forest estate is its very skewed age-class distribution, which is greatly amplified by the artificial separation of the whole estate into pre-1900 and post-1990 stands. Exotic forests are usually harvested at about 25–30 years of age. As a consequence, carbon stocks of the estate undergo large fluctuations over time scales of Commitment Periods (see Chapter 3). Hence, under 'Post-1990 Gross-Net' accounting, the large sink of the estate over CP1 and CP2 largely disappears by CP3 and would even become a source over CP4 and CP5. These fluctuations would not occur under the ACS because once the estate reaches the average carbon stocks that are typical for a particular land use/management combination, further fluctuations are not counted unless any management changes lead to changes in average carbon stocks. While the exotic forest estate remains under that land use, however, it is counted as carbon neutral once carbon stocks have ramped up to the average. Application of a forward-looking baseline would similarly smooth

any fluctuations from the carbon-stock changes of the existing estate so that only new plantings would generate credits.

4.4 Mitigation Options and Their Encouragement Under Different Accounting Options

There is significant potential for improvement of New Zealand's net position by the re-establishment of forests on pasture land and the prevention of further deforestation. Numerically over a Commitment Period, there is similar potential by planting 40 000 ha yr⁻¹ of exotic plantations, establishing 100 000 ha yr⁻¹ of indigenous forests or stopping the deforestation of 10 000 ha yr⁻¹ of exotic plantations. Deforestation makes a large contribution per hectare because large carbon changes (are assumed to) occur instantaneously, whereas forest regrowth is a much slower process. This factor is largely negated by the fact that, once planted, forests can make contributions over several Commitment Periods, whereas deforestation primarily has a short-term effect immediately after it occurs. Inclusion of harvested wood products would lessen the sharpness of the carbon release upon deforestation.

Afforestation with indigenous and exotic forests differs in their contribution per ha because exotic forests tend to have much higher growth rates, partly because they tend to be planted on better land and receive better site preparation, weed control and fertiliser additions. This difference in sequestration rates over a few Commitment Periods is also negated by the fact that exotic forests tend to be harvested after 25–30 years and lose much of their carbon stocks, whereas indigenous forests can continue to grow and accumulate carbon for much longer periods.

Different carbon accounting options differently encourage, discourage or are neutral in terms of positive mitigation options. The four options, 'All-forest gross-net', 'All-forest net-net', 'Forward-looking baseline' and ACS, all similarly encourage any actions with any positive carbon-mitigation outcomes. However, 'Post-1990 gross-net' and 'All-lands gross-net with cap' provide incentives only for the maintenance of carbon stocks in the post-1990 estate, while ignoring carbon stocks in the pre-1990 estate. For example, it would currently be advantageous to harvest pre-1990 forests while extending the rotation length of post-1990 stands. It is obvious that such management shifts would have no beneficial outcomes on atmospheric CO₂ concentrations and, in fact, might even have detrimental consequences.

4.5 Soil Carbon Measurements and Implications

Changes in carbon stocks are dominated by changes in biomass carbon but recognised soil-carbon stock changes make a quantitatively significant contribution to overall changes. Furthermore, there may be other changes in soil carbon that are either not currently recognised or may even be incorrectly assigned. These problems stem from past use and application of measurement methodologies that are inappropriate for carbon accounting purposes. The two principal problems are measurements to shallow depth and a failure to correctly deal with bulk density.

For carbon accounting, all C-stock changes ultimately matter irrespective of the depth at which they occur. Measurements down to 5 or 10 cm might simply miss changes that occur deeper in the soil or at least assess them quantitatively incorrectly. Problems are particularly severe if there is a change in the vertical distribution of soil carbon. This is very likely when

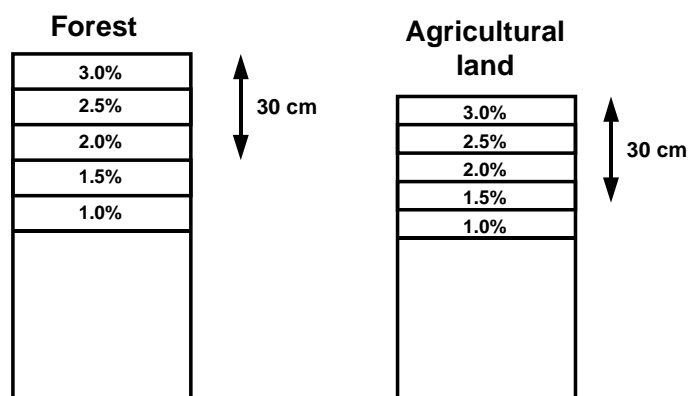
soil cultivation is involved, given that cultivation redistributes soil carbon within the depth of the plough layer. Hence, it appears that the beneficial effect of zero tillage might have been overestimated because the shallow sampling on which much of the current knowledge is based would have missed this vertical redistribution of soil carbon and would not have recognised that soil carbon build-up near the surface under zero tillage might have been partly compensated by some carbon losses in deeper soil layers.

The IPCC default of 30 cm is thus a minimum depth to which sampling should be carried out, but even this might miss changes occurring deeper in the soil, as appears to be the case in soil carbon losses under dairy pastures (e.g., Schipper et al. 2007). It is true that soil carbon changes are typically more pronounced at shallower depths because the older and more recalcitrant soil carbon at greater depths often tends to be slower to respond to changed conditions. However, it would make more sense to avoid artificially setting limits (such as 30 cm) as the appropriate sampling depth but instead try to ascertain to what depth changes are actually occurring in response to specific land-use /management changes and use such observations as guides for sampling protocols.

An even more serious problem arises with respect to bulk density changes and the use of a fixed analysis depth. Many land use/management changes lead to changes in bulk density. For instance, Murty et al. (2002), when reviewing the literature of relevant deforestation studies, found that bulk density increased by an average of 13% over about 10 years when forests were cleared. In some studies, increases of as much as 60% were observed. Such changes in bulk density then lead to calculated changes in soil-carbon stocks when they are summed to a fixed depth, even in situations without any carbon actually lost or gained by the system as a whole (Figure 1). Calculated proportional changes in carbon stocks are typically less than the proportional changes in bulk density because carbon concentrations typically decrease with depth so that the inclusion or exclusion of extra layers of soil around 30 cm leads to a less than proportional change in overall carbon stocks or average concentration. Nonetheless, calculated errors can still be substantial. In the example in Figure 1, this could amount to an apparent spurious C loss of 15 t C ha⁻¹ or more.

For carbon accounting purposes it is therefore not appropriate to calculate carbon stocks to a fixed depth. Instead, it is strongly recommended that stocks should be calculated to a variable depth that corresponds to the same amount of mineral soil (e.g., Ellert et al. 2000). While this procedure is more difficult to operationalise, it is regarded as the only method that can avoid true carbon-stock changes being confounded with apparent changes that are entirely due to bulk density changes.

Figure 1 Effect of a notional change in bulk density with land conversion on calculated carbon stocks or carbon concentration. This notional example gives a large change in bulk density by 33% in moving from forest to agricultural land but without any change in actual carbon stocks. However, the bulk density change would lead to a 20% increase in calculated carbon stocks and a 10% decrease in average carbon concentration for calculations to 30 cm.



5. Key Information Gaps and Overall Prioritised Research Plan

It is critically important for New Zealand to have a thorough appreciation of the effect of different accounting options on accountable net emissions. Various components of the national carbon budget and its response to land-use and management changes are still only incompletely understood, and further research is warranted to address these gaps. However, the overall LULUCF budget is dominated by biomass carbon changes, and these current and future trends are generally fairly well understood and documented. Here, too, further refinement is warranted, but the most important components are well enough known.

There is, however, a need for further work on using that information for the assessment of different accounting options. To some extent this has been done both in Chapter 3 of this report, where a detailed numeric assessment was applied to a range of specific accounting options, and in this Chapter, where the numeric information was used more generically to assess a wider range of accounting options. In order for New Zealand to be better and more fully able to assess the implication of the range of possible accounting options, it is advisable to conduct further targeted research to assess the implications of different accounting options on New Zealand's net position.

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APPENDIX 1: Datasets—Forest Management And Soil Carbon

1.1 OVERVIEW: FOREST MANAGEMENT AND SOIL CARBON

Datasets associated with the studies shown below are summarised in this appendix.

Sectors	Information
Exotic Forestry	<p>Datasets relating to the effects of exotic plantation forest management</p> <ol style="list-style-type: none"> 1. Tarawera (Residue Management Effects) 2. Rotoehu (Cultivation Effects – Spot-mounding) 3. Lochinver (Cultivation Effects – Line-ripping) 4. Mahurangi (Forest Harvesting Effects) 5. Puruki (Forest Harvesting Effects) 6. LTSP Series I Trials (Residue Management and Fertiliser Effects) 7. LTSP Series II Trials (Compaction, Fertiliser, Species and Herbicide Effects) 8. LTSP Series III Trials (Species Effects) 9. Tiketere Agroforestry (Tree Stocking Effects) 10. Kyoto Forest Inventory Plots (Age Class Effects) 11. Balmoral-Waitaki (Tree Stocking Effects) 12. Orton Bradley (Species Effects) 13. Boron trial series (Herbicide Effects) 14. Mid-rotation N and P trials (Fertiliser Effects) <p>New datasets relating to the effects of ARD</p> <ol style="list-style-type: none"> 1. Balmoral-Waitaki (Afforestation Effects) 2. Orton Bradley (Afforestation Effects) 3. Afforested LTSP III (Afforestation Effects)

1.2 DESCRIPTION OF DATASETS RELATING TO THE EFFECTS OF EXOTIC PLANTATION FOREST MANAGEMENT

1.2.1 Tarawera (Residue Management Effects)

SUMMARY

This an experimental dataset collected to determine the impacts of different levels of harvest residue retention on soil C stocks in an exotic plantation forest system (*P. radiata* forest on Typic Tephric Recent Soils in scoria). The forest floor materials and mineral soils were sampled to a depth of 30 cm. The data is held by Scion in an Excel spreadsheet. Removal of both harvest residues and forest floor horizons resulted in significantly lower C stocks (by up to 10 Mg ha⁻¹) in some pools (0-0.1 m mineral soil, forest floor, and total soil) than residue retention at mid-rotation, but only with the inclusion of mineral soil coarse fraction stocks.

DATABASE DESCRIPTION

Name of the dataset:	Tarawera (Residue Management Effects).
Primary contact:	Dr Haydon Jones, Scion.
Data ownership / accessibility:	Scion, MFE / Publicly available.
Dataset type:	Experimental
Data storage:	Data is currently held in an Excel spreadsheet on a network drive accessible by Carbon Team staff.
Broad Land use / Landform sampled:	Exotic plantation forestry (<i>P. radiata</i>) / Flood plain
Number of sites:	One trial site (Tarawera LTSP I) employing a split-plot, randomised-block design comprising four blocks (all contained within a ~ 1 ha area)
Geographical spread:	Trial located in the Bay of Plenty, near Kawerau
Soil orders:	Recent Soils
Depth of sampling: 0-30 cm	Yes
Specific sampling depths:	Forest floor materials were sampled by horizon: L (needles, bark, and cone) and FH horizons. Mineral soil was sampled from the following depth ranges: 0-10, 10-20, 20-30 cm.
Bulk Density measurements:	Yes (for every individual sample).
Multiple sampling through time:	No
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	GPS coordinates for a point within the trial site is available
Soil description:	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	Yes (with treatment in some pools)
Land use history recorded:	Yes
If Yes above, then history for what period?	44+ yrs preceding last sampling and up to present.
Management history available:	

History timeframe:	The trial has been under exotic plantation forestry (<i>P. radiata</i>) for about the last 46 years. Prior to this, the land use at the site is thought to be pastoral agriculture.
Management factors held:	Specific information held on harvest residue management treatments (core treatments were: whole-tree harvest, stem-only harvest, and whole-tree harvest + forest floor removal) and fertiliser rates applied (fertilised plots received N, P, K, Mg, B, and Cu fertilisers at various rates and times but <i>note that only unfertilised plots sampled for dataset</i>), and on other silvicultural manipulations (initial stocking was about 100 stems per plot but was thinned-to-waste at ages 5 and 11, trees were not pruned, weeds were controlled).
Data held in what form:	Ancillary data regarding the trial is variously held in both electronic form on a network drive and in trial notebooks (held and maintained by Graeme Oliver, Doug Graham, and Stephen Pearce).

Mitigation opportunities to minimise or improve soil C:

Findings based on this dataset suggested there are opportunities for soil C loss mitigation via the retention of both harvest residues and forest floor materials on-site.

Supplementary information on soils:

Total N (% and stocks) data are also included in this particular dataset. Stocks of C contained in the coarse (> 2 mm) fraction were measured to assess the size of this pool in these soils. A range of tree growth, above-ground biomass, woody decay, and soil property data also exists in various other datasets relating to this trial.

Associated publications:

Jones, H.S., Garrett, L.G., Beets, P.N., Kimberley, M.O., Oliver, G.R. Impacts of harvest residue management on soil carbon stocks in a plantation forest. *Soil Science Society of America Journal*, In press.

Jones, H.S. 2007. Impacts of harvest residue management on soil carbon stocks in Tarawera Forest – updated and revised results. Ensis contract report prepared for the Ministry for the Environment.

Free-form database description

Origin:

This dataset was collected primarily to establish the impacts of three different harvest residue management treatments (whole-tree harvest, stem-only harvest, and whole-tree harvest + forest floor removal) on soil C stocks but also contributed to a wider assessment of C stocks in plantation forest systems in NZ.

Collection methodology:

A full description of the site, trial design, and methodology used is given in Jones *et al.* (In Press).

A split-plot, randomized block design incorporating three main harvest residue management treatments was used in the establishment of the trial. Each plot was 60 × 30 m in size and contained two split-plots (fertilized and unfertilized) of 20 × 20 m treated area with each surrounded by a 5 m buffer. Four blocks were installed giving four fertilized and four unfertilized replicate plots of each of the three main treatments in total. Only unfertilized plots were sampled in this study. All plots initially contained about 100 trees but were thinned-to-waste at age 5 (in 1994) and again at age 11 (in 2000). The trees were not pruned.

Within each unfertilized plot, a total of nine sample points were located on an 8 × 8 m regular grid pattern. Forest floor materials (needle litter and fermented/humic material) and mineral soils (0-0.3 m) were sampled at each of the nine points. The sampling was undertaken in two separate phases. In September 2005 (stand aged 16 years), the forest floor was sampled at all points whereas the mineral soil was sampled from a minimum of five points in each plot. The second sampling occurred in September 2006 (stand aged 17 years) and involved the collection of additional mineral soil samples to give a total of nine points sampled within each plot. The additional mineral soil samples were collected to help reduce within-plot variability.

Two forest floor horizons were identified and sampled: litter (L) and fermented/humic (FH) horizons. For the purposes of this study, the L horizon included needle, bark, and cone materials only. Branch materials were collected separately and were not included in the analyses presented here because the branch material was largely derived from thinned stems (second thinning) and were, therefore, unlikely to reflect any effect of the harvest residue management treatments on mineral soil C. It is likely that almost all the material collected as L accumulated after harvesting and replanting. Prior to mineral soil sampling, the L horizon was sampled from within a 0.25 m² metal quadrat centered on the sample point. The FH material was then collected from within a sampling ring (of 98 mm internal diameter) inserted in the centre of the same 0.25 m² area. The L and FH horizon thicknesses were recorded during sampling. All L and FH materials contained within their respective sampling areas were collected.

The very gravelly nature of the subsoil at Tarawera precluded the use of small-diameter tube samplers – often employed for the rapid collection of large numbers of surface soil samples – for sample collection to 0.3 m depth due to the likelihood of tube blockage and incomplete sample recovery. Therefore, the mineral soil was collected from three 0.1 m depth ranges (0-0.1, 0.1-0.2, and 0.2-0.3 m) using a 98 mm internal diameter (× 100 mm depth) stainless-steel soil sampling ring positioned in the centre of the same 0.25 m² area immediately after the collection of the FH material. Volumetric soil-core sampling allowed for the direct

measurement of the bulk density (D_b ; g cm^{-3}) of each soil sample collected in addition to providing a sub-sample for C and N analysis. The D_b data were used to express

Sample analysis:

For a detailed description see Jones et al. (In Press). Mineral soil and forest floor samples were air-dried before being analysed for total C and N using a LECO FPS-2000 CNS thermal combustion furnace.

Strengths and limitations:

The strengths of this dataset are that C stocks were measured in 10 cm increments to 30 cm, that bulk density was measured for every individual soil sample, and that treatment effects were tested using a robust and proven experimental design. Some limitations of the dataset are that the results relate to a single point in time following harvesting (mid-rotation) and to a specific soil type (Recent Soils formed in scoria) and the change detected is relative rather than absolute. Also, greater sample numbers may have allowed the detection of further significant effects of treatments but numbers were limited by the costly nature of the methodology required for sampling these soils.

Assumptions and uncertainties:

It is assumed that all the treatments were applied accurately and consistently over all replicate plots and that no mineral soil was removed or physically disturbed by treatment application. It remains uncertain as to whether residue removal alone could be sufficient to result in lower soil C stocks than residue retention (common practice). The results of the study have highlighted the potential importance of measuring C stocks in the coarse fractions of mineral forest soils formed in vesicular parent materials — possible implications for future stock assessment methodology.

Main findings relevant to change in soil C stocks:

The main findings of the work are summarised as follows:

1. No significant impacts on stocks of C and N were observed in the mineral soil fine fraction. However, the inclusion of the coarse fraction stocks (representing 25% of total mineral soil C to 0.3 m depth) enabled the detection of significant treatment effects.
2. Stem-only harvesting (residue retention) had significantly larger C stocks in the 0-0.1m total mineral soil (fine + coarse fractions), forest floor (L + FH), and total soil (forest floor + 0-0.3 m total mineral soil) pools than whole-tree harvesting (residue removal) plus forest floor removal (Figure 1.1).

Future plans:

A similar dataset is currently being developed to examine the impacts of harvest residue management and fertiliser use on soil C stocks at a different site – on Pumice Soils in Kinleith forest – and a comparison of the results is intended.

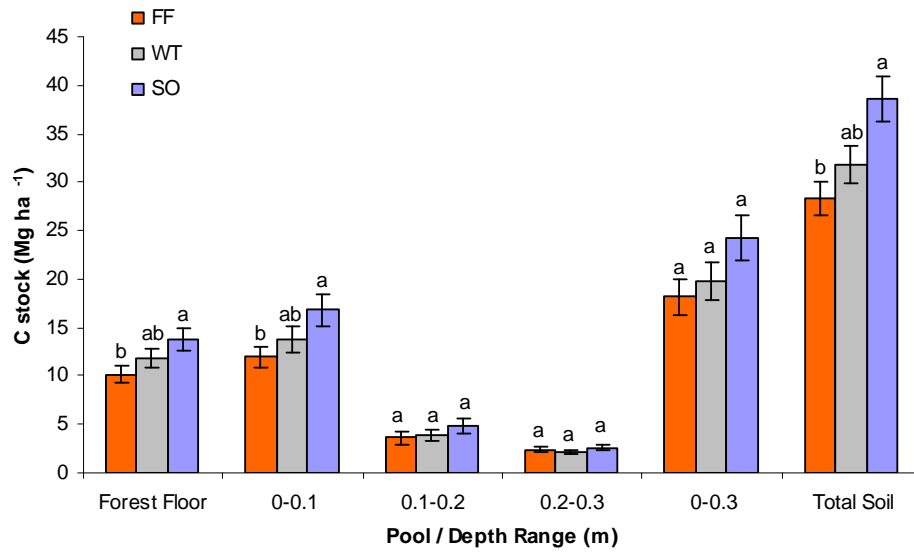


Figure 1.1. Mean C stocks in total mineral soil, combined forest floor (L + FH), and total soil (forest floor + total mineral soil) pools under forest floor (FF), whole tree (WT) and stem only (SO) treatments. Geometric means presented. Means carrying the same letter are not significantly ($P < 0.05$) different. Means are based on four replicate values per treatment. Error bars represent the standard error about the mean.

1.2.2 Rotoehu (Cultivation Effects – Spot mounding)

SUMMARY

This an experimental dataset collected to determine the impacts of spot-mounding cultivation on soil C stocks in an exotic plantation forest system (*P. radiata* forest on Vitric Orthic Allophanic Soils and Allophanic Orthic Pumice Soils). The mineral soil was sampled to a depth of 30 cm. The data is held by Scion in an Excel spreadsheet. Spot-mounding cultivation resulted in a net total (area-adjusted) loss of about 4 Mg C ha⁻¹ from the top 30 cm of mineral soil at Rotoehu.

DATABASE DESCRIPTION

Name of the dataset:	Rotoehu (Cultivation Effects - Spot mounding).
Primary contact:	Dr Haydon Jones, Scion.
Data ownership / accessibility:	Scion, MFE / publicly available.
Dataset type:	Experimental
Data storage:	Data is currently held in an Excel spreadsheet on a network drive accessible by Carbon Team staff.
Broad Land use / Landform sampled:	First rotation, young, exotic plantation forestry (<i>P. radiata</i>) / Aggraded floodplain (paleochannel) and a more elevated terrace.
Number of sites:	One trial site (Rotoehu LTSP III) comprising four radiata plots, each subdivided into four sub-plots.
Geographical spread:	Trial located in the Pongakawa Valley, Bay of Plenty.
Soil orders:	Allophanic Soils (~ 30 % of data points) and Pumice Soils (~ 70 % of data points)
Depth of sampling: 0-30 cm	Yes
Specific sampling depths:	Mineral soil 0-10, 10-20, 20-30 cm.
Bulk Density measurements:	Yes (for every individual sample).
Multiple sampling through time:	No
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	GIS shape-files giving trial boundaries; GPS coordinates (to be confirmed)
Soil description:	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	Yes
Land use history recorded:	Yes
If Yes above, then history for what period?	1+ yrs preceding last sampling and up to present.
Management history available:	
History timeframe:	The trial was planted about 3 years ago and the previous land use was agroforestry (grazed pasture with sparse poplar trees).
Management factors held:	Soils at the site were cultivated using the spot-mounding technique prior to planting. Various trial maintenance activities such as weed control are recorded.

Data held in what form: Limited management history data is held in electronic trial information sheets and other files on a network drive.

Mitigation opportunities to minimise or improve soil C:

Findings based on this dataset suggested that opportunities for soil C loss mitigation may arise through either the avoidance of forest soil cultivation (where possible) or the use of a cultivation technique that causes less soil disturbance than spot-mounding (e.g. line-ripping).

Supplementary information on soils:

Total N (% and stocks) data is also included in this particular dataset.

Associated publications:

Jones, H.S., 2007. Effects of forest soil cultivation on the carbon stocks and other properties of contrasting volcanic soils. Contract report prepared for the Ministry for the Environment. Contract 03/04 – 0271 – L. June 2007.

Free-form database description

Origin:

This dataset was collected to establish the effects of a specific type of forest soil cultivation (spot-mounding) on soil C stocks at a first-rotation plantation forest site.

Collection methodology:

A full description of the site, trial design, and methodology used is given in Jones (2007).

Four radiata plots, approximately 1500 m² in area, were used in this study. Each plot contained 13 rows (mound-and-hollow sequences) with 10 spot-mounds in each row. Mounds were spaced 2.5 m apart and rows were spaced 5 m apart, leaving 12 strips of undisturbed soil within each plot. In total, 130 spot-mounds and 210 hollows occurred within each plot. Each plot was subdivided into quarters, creating four sub-plots per plot to ensure an even spatial representation of sample points and an adequate level of replication. Within each sub-plot the locations of three clusters of samples (each containing three individual sample points) were randomly selected. Two disturbance features associated with spot-mounding were identified at Rotoehu (spot-mounds and hollows). A sample-point cluster consisted of one point on a spot-mound, one in an adjacent hollow, and the other point in the centre of an adjacent undisturbed strip. A cluster-sampling approach was adopted to ensure that the comparison of soil property values among disturbance features and undisturbed soil was statistically valid and sufficiently robust. In summary, three sample points were located on each disturbance feature and the undisturbed soil within each sub-plot which equates to 12 points per plot and a total of 48 points across the site at Rotoehu.

At each sample point, the mineral soil was sampled at three depths ranges (0-0.1, 0.1-0.2, and 0.2-0.3 m) using a 98 mm internal diameter (× 100 mm depth) soil sampling ring. Samples were collected perpendicular to the soil surface in all cases (e.g. perpendicular to the sloping sides of spot- and rip-mounds). Sampling was undertaken in October 2006; approximately 15 months after spot-mounding.

The areas of land on the horizontal plane occupied by each soil disturbance feature (i.e. spot-mounds and hollows) were calculated at the sub-plot level and were converted to proportions

of the total sub-plot area. These data were collected for use in the assessment of net total C and N stock change under forest soil cultivation.

Sample analysis:

Mineral soil samples were air dried at $\leq 40^{\circ}$ C to constant weight (and total air-dry weight recorded) before being passed through a 2 mm sieve to separate the fine (< 2 mm) and coarse (> 2 mm) fractions. Air-dry weights of the fine fraction were recorded and the coarse fraction materials were weighed and stored for future analysis. Each fine fraction sample was split in two, with half being oven dried at 105° C (to constant weight) to provide representative moisture factor values (for D_b calculation), the remainder being ground to < 1 mm using a rock-mill in preparation for C and N analysis. Samples were bulked by sub-plot (on a mass-weighted basis) giving a site total of 16 samples per disturbance feature per depth range for the measurement of total C and N (Table 1). However, the fine fraction D_b was determined for each individual sample before being averaged (ratio of total mass over total volume) by sub-plot for statistical analysis. Bulk samples were analysed for total C and N using a LECO FPS-2000 CNS thermal combustion furnace (LECO Corp., St Joseph, MI).

Data from samples taken on spot-mounds (small sloping features) were corrected for slope.

A method for the quantitative assessment of net cultivation-induced changes in the C and N stocks of plantation forest soils was developed. The method can be described as an area-adjusted approach that compares pre-cultivation stocks with post-cultivation stocks over the entire site. The difference between the pre- and post-cultivation stocks gives the magnitude and direction of change. In this study the comparisons were made at the sub-plot level. The method requires quantitative information relating to the proportion of a given land area (on the horizontal plane) occupied by each identified disturbance feature. The area of undisturbed soil can be obtained by difference. It was assumed that stocks measured in the undisturbed soil were representative of and equivalent to the pre-cultivation stocks across the whole area in question. The stocks of each disturbance feature and undisturbed soil were scaled by the relative proportion of the total area each occupied and the sum of these 'area-adjusted' stocks was taken as the total post-cultivation stock for the area in question.

Strengths and limitations:

The strengths of this dataset are that C stocks were measured in 10 cm increments to 30 cm, that bulk density was measured for every individual soil sample, and that treatment effects were tested using a robust experimental design. Furthermore, undisturbed soils were sampled in addition to cultivated, which allowed for the assessment of absolute change in C stocks. A method was developed with this dataset to determine net total (area-adjusted) soil C stock change in cultivated plantation forest lands. Some limitations of the dataset are that the results relate to a single point in time following cultivation and to only two soil types found within a single site (Allophanic and Pumice Soils). That is, the findings may be very time- and site-specific.

Assumptions and uncertainties:

It was assumed that most of the C stock loss observed from the top 30 cm of mineral soil was to the atmosphere, rather than redistributed below 30 cm under spot-mounds. An assessment of soil profile morphology in spot-mounds tended to support this assumption (little evidence of significant topsoil redistribution below 30 cm). However, stocks below 30 cm in mounds were not quantified. Therefore, some uncertainty still surrounds the ultimate fate of the C lost due to spot-mounding. It was also assumed that C stocks calculated for undisturbed soils (in

inter-row strips) were representative of pre-cultivation stocks for the entire area under cultivation.

Main findings relevant to change in soil C stocks:

The main findings of the work in relation to soil C stocks were that:

1. In comparison to undisturbed soil, stocks of C in hollows were significantly lower (by 30 Mg ha⁻¹) in all depth ranges whereas stocks of C in spot-mounds were significantly lower (by 10 Mg ha⁻¹) in the top 10 cm only (Figure 1.2).
2. Spot-mounding cultivation resulted in a net total (area-adjusted) loss of more than 4 Mg C ha⁻¹ in the top 30 cm of mineral soil at Rotoehu. Most of this C was lost from the top 10 cm (Figure 1.3). There is little (or no) evidence to suggest that C was redistributed down the profile. Therefore, it is thought that most was lost to the atmosphere.

Future plans:

We plan to submit a manuscript on based on this dataset to a referred journal later this year (2008) and to undertake further research into the impacts of forest soil cultivation practices.

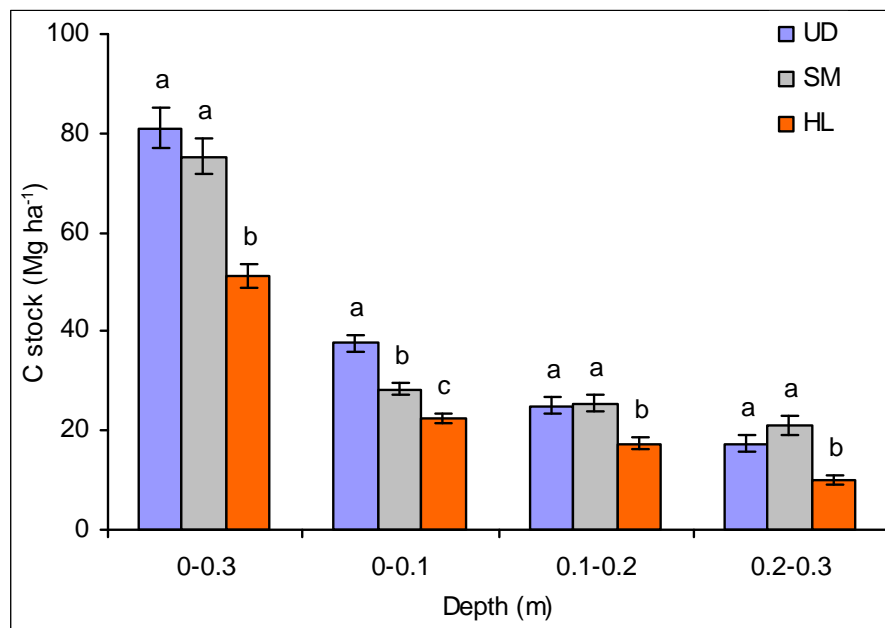


Figure 1.2. Mean C stocks in hollows (HL), spot-mounds (SM), and undisturbed (UD) soil after spot-mounding at Rotoehu. Geometric means are presented. Within each depth range, means carrying the same letter are not significantly ($P < 0.05$) different. Error bars represent the standard error about the mean.

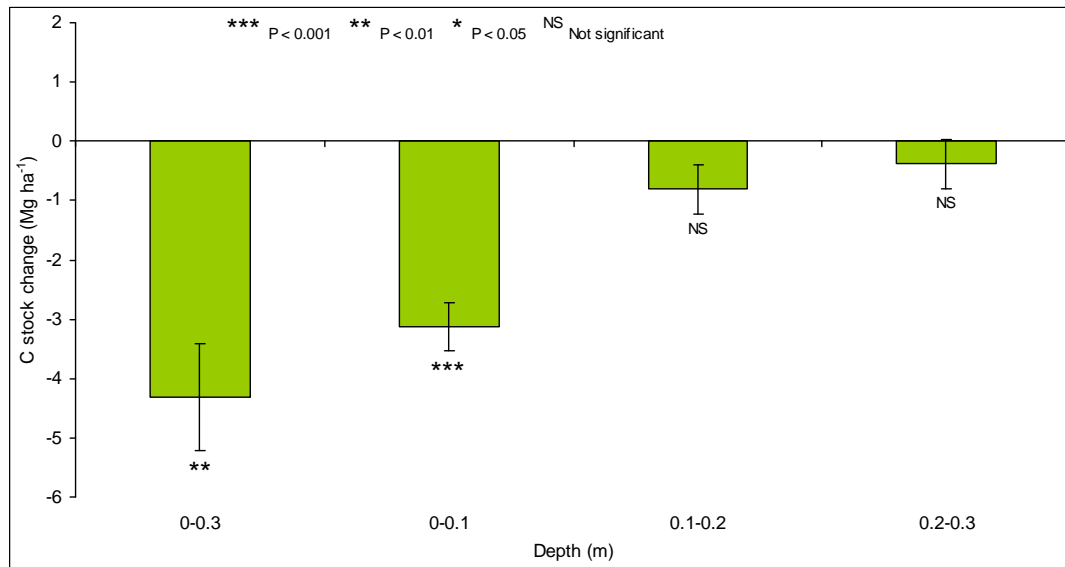


Figure 1.3. Mean change in area-adjusted C stocks after spot-mounding at Rotoehu. Arithmetic means are presented. Error bars represent the standard error about the mean.

1.2.3 Lochinver (Cultivation Effects – Line-ripping)

SUMMARY

This an experimental dataset collected to determine the impacts of line-ripping cultivation on soil C stocks in an exotic plantation forest system (*P. radiata* forest on Immature Orthic Pumice Soils). The mineral soil was sampled to a depth of 30 cm. The data is held by Scion in an Excel spreadsheet. Line ripping resulted in no significant net total (area-adjusted) change in C stocks of the top 30 cm of mineral soil at Lochinver.

DATABASE DESCRIPTION

Name of the dataset:	Lochinver (Cultivation Effects - Line-ripping).
Primary contact:	Dr Haydon Jones, Scion
Data ownership / accessibility:	Scion, MFE / publicly available
Dataset type:	Experimental
Data storage:	Data is currently held in an Excel spreadsheet on a network drive accessible by Carbon Team staff.
Broad Land use / Landform sampled:	Land use is second-rotation exotic plantation forestry (<i>P. radiata</i>) / Site sampled was situated on an essentially flat outwash plain within the central plateau.
Number of sites:	One trial site (Lochinver LTSP III) comprising four radiata plots, each subdivided into four sub-plots.
Geographical spread:	Trial located near Rangitaiki on the Napier-Taupo Highway, central North Island.
Soil orders:	Pumice Soils
Depth of sampling: 0-30 cm	Yes
Specific sampling depths:	Mineral soil 0-10, 10-20, 20-30 cm.
Bulk Density measurements:	Yes (for every individual sample).
Multiple sampling through time:	No
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	GIS shape-files giving trial boundaries and GPS coordinates (to be confirmed)
Soil description:	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	No (no net change over the 0-30 cm pool)
Land use history recorded:	Yes
If Yes above, then history for what period?	33+ yrs preceding last sampling and up to present.
Management history available:	
History timeframe:	The trial was planted about 5 years ago and the previous land use was first-rotation exotic plantation forestry. We hold information on management history over this 5-year period but the land owners may hold information on

Management factors held:	the management history of the first rotation and perhaps the preceding land use.
Data held in what form:	Soils at the site were cultivated using the line-ripping technique prior to planting. Various trial maintenance activities such as weed control are recorded. Limited management history data is held in electronic trial information sheets and other files on a network drive.

Mitigation opportunities to minimise or improve soil C:

Findings based on this dataset suggested that opportunities for soil C loss mitigation may arise, at least on relatively coarse-textured and infertile soils, through the use of line-ripping in preference to a cultivation technique that causes greater soil disturbance (e.g. spot-mounding).

Supplementary information on soils:

Total N (% and stocks) data is also included in this particular dataset.

Associated publications:

Jones, H.S., 2007. Effects of forest soil cultivation on the carbon stocks and other properties of contrasting volcanic soils. Contract report prepared for the Ministry for the Environment. Contract 03/04 – 0271 – L. June 2007.

Free-form database description

Origin:

This dataset was collected to establish the effects of a specific type of forest soil cultivation (line-ripping) on soil C stocks at a second-rotation plantation forest site.

Collection methodology:

A full description of the site, trial design, and methodology used is given in Jones (2007).

Four radiata plots, about 2100 m² in area, were used in this study. Each plot contained 10 continuous rip-mounds spaced 5 m apart leaving 10 intervening strips of undisturbed soil. Rip-mounds had an average length of approximately 39 m in all plots. Each plot was subdivided into quarters, creating four sub-plots per plot to ensure an even spatial representation of sample points and an adequate level of replication. Within each sub-plot at both sites the locations of three clusters of samples (each containing three individual sample points) were randomly selected. A sample-point cluster consisted of two points on a rip-mound (each situated in the centre of an unplanted space either side of a seeding) and the other point in the centre of an adjacent undisturbed strip. A cluster-sampling approach was adopted to ensure that the comparison of soil property values among disturbance features and undisturbed soil was statistically valid and sufficiently robust. In summary, three sample points were located on each disturbance feature and the undisturbed soil within each sub-plot which equates to 12 points per plot and a total of 48 points across the site.

At each sample point, the mineral soil was sampled at three depths ranges (0-0.1, 0.1-0.2, and 0.2-0.3 m) using a 98 mm internal diameter (× 100 mm depth) soil sampling ring. Samples were collected perpendicular to the soil surface in all cases (e.g. perpendicular to the sloping

sides of rip-mounds). Sampling was undertaken in October 2006; approximately 38 months after continuous ripping.

The areas of land on the horizontal plane occupied by each soil disturbance feature (i.e. rip-mounds) were calculated at the sub-plot level and were converted to proportions of the total sub-plot area. These data were collected for use in the assessment of net total C and N stock change under forest soil cultivation.

Sample analysis:

Mineral soil samples were air dried at $\leq 40^{\circ}$ C to constant weight (and total air-dry weight recorded) before being passed through a 2 mm sieve to separate the fine (< 2 mm) and coarse (> 2 mm) fractions. Air-dry weights of the fine fraction were recorded and the coarse fraction materials were weighed and stored for future analysis. Each fine fraction sample was split in two, with half being oven dried at 105° C (to constant weight) to provide representative moisture factor values (for D_b calculation), the remainder being ground to < 1 mm using a rock-mill in preparation for C and N analysis. Samples were bulked by sub-plot (on a mass-weighted basis) giving a site total of 16 samples per disturbance feature per depth range for the measurement of total C and N (Table 1). However, the fine fraction D_b was determined for each individual sample before being averaged (ratio of total mass over total volume) by sub-plot for statistical analysis. Bulk samples were analysed for total C and N using a LECO FPS-2000 CNS thermal combustion furnace (LECO Corp., St Joseph, MI).

Data from samples taken on rip-mounds (small sloping features) were corrected for slope.

A method for the quantitative assessment of net cultivation-induced changes in the C and N stocks of plantation forest soils was developed. The method can be described as an area-adjusted approach that compares pre-cultivation stocks with post-cultivation stocks over the entire site. The difference between the pre- and post-cultivation stocks gives the magnitude and direction of change. In this study the comparisons were made at the sub-plot level. The method requires quantitative information relating to the proportion of a given land area (on the horizontal plane) occupied by each identified disturbance feature. The area of undisturbed soil can be obtained by difference. It was assumed that stocks measured in the undisturbed soil were representative of and equivalent to the pre-cultivation stocks across the whole area in question. The stocks of each disturbance feature and undisturbed soil were scaled by the relative proportion of the total area each occupied and the sum of these 'area-adjusted' stocks was taken as the total post-cultivation stock for the area in question.

Strengths and limitations:

The strengths of this dataset are that C stocks were measured in 10 cm increments to 30 cm, that bulk density was measured for every individual soil sample, and that treatment effects were tested using a robust experimental design. Furthermore, undisturbed soils were sampled in addition to cultivated, which allowed for the assessment of absolute change in C stocks. A method was developed with this dataset to determine net total (area-adjusted) soil C stock change in cultivated plantation forest lands. Some limitations of the dataset are that the results relate to a single point in time following cultivation and to only one soil type found within a single site (Immature Orthic Pumice Soils). That is, the findings may be very time- and site-specific.

Assumptions and uncertainties:

It was assumed that C stocks calculated for undisturbed soils (in inter-row strips) were representative of pre-cultivation stocks for the entire area under cultivation. This it thought to be a valid assumption provided there strips were not significantly compacted during the cultivation process.

Main findings relevant to change in soil C stocks:

The main findings of the work in relation to soil C stocks were that:

1. Stocks of C in rip-mounds were not significantly different from those in undisturbed soil within the 0-30 cm depth range because the significantly lower stocks (by 9 Mg C ha⁻¹) observed in rip-mounds in the top 10 cm were partly offset by significantly greater stocks (by 4 Mg C ha⁻¹) in rip-mounds in the 20-30 cm depth range. This result clearly indicates that some redistribution of C down the profile occurred after line-ripping at the Lochinver site (Figure 1.4).
2. Line-ripping resulted in a significant net total (area-adjusted) loss of 2.5 Mg C ha⁻¹ from the top 10 cm of mineral soil. This was off-set by a significant gain of 1.1 Mg C ha⁻¹ in the 20-30 cm depth range. Therefore, no significant change in the C stocks of the top 30 cm was observed after line-ripping at Lochinver (Figure 1.5).

Future plans:

We plan to submit a manuscript on based on this dataset to a referred journal later this year (2008) and to undertake further research into the impacts of forest soil cultivation practices.

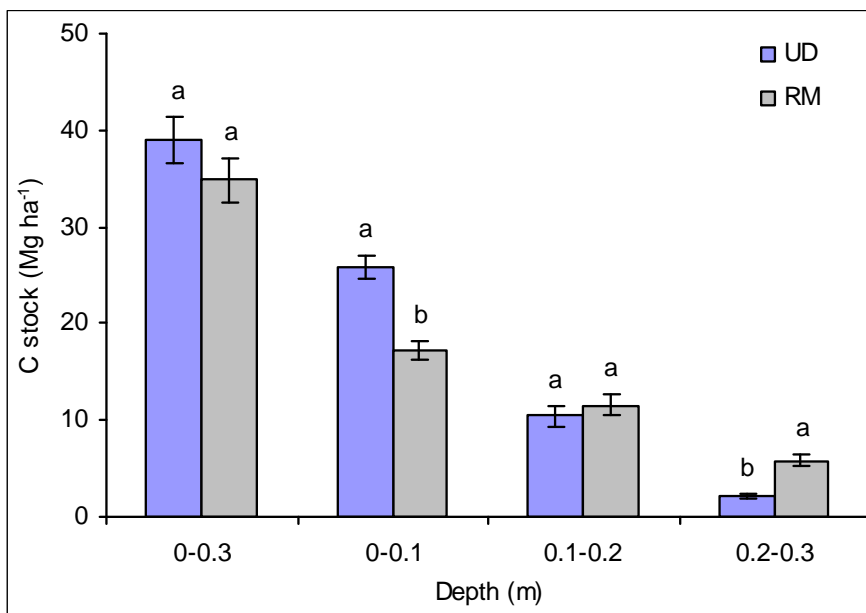


Figure 1.4. Mean C stocks in rip-mounds (RM) and undisturbed (UD) soil after ripping at Lochinver. Back-transformed means are presented. Within each depth range, means carrying the same letter are not significantly ($P < 0.05$) different. Error bars represent the standard error about the mean.

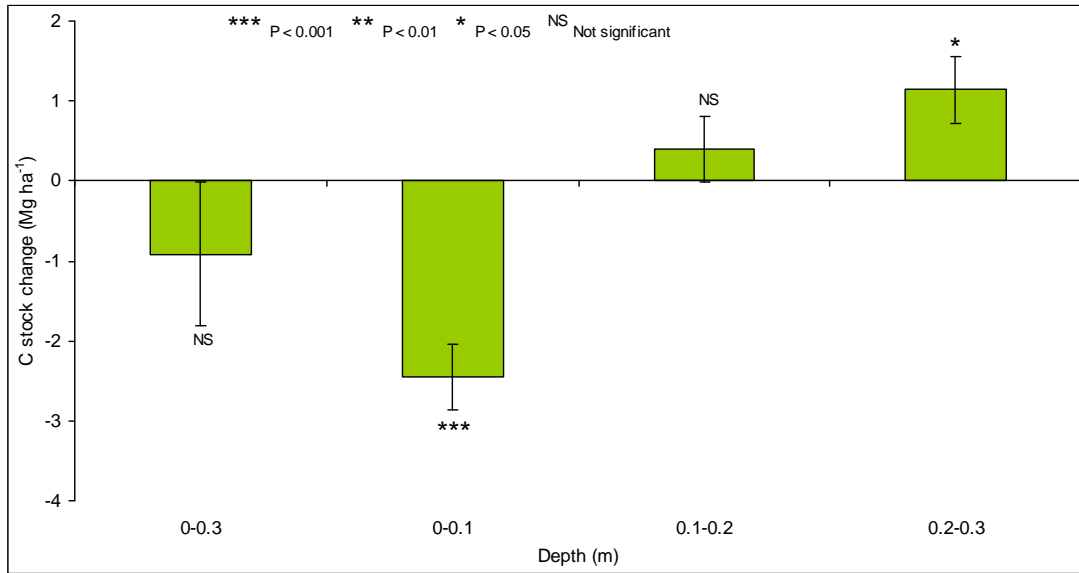


Figure 1.5. Mean change in area-adjusted C stocks after line-ripping at Lochinver. Arithmetic means are presented. Error bars represent the standard error about the mean.

1.2.4 Mahurangi (Forest Harvesting Effects)
 DATABASE DESCRIPTION

Name of the dataset:	Mahurangi (Forest Harvesting Effects).
Primary contact:	Dr Haydon Jones, Scion
Data ownership / accessibility:	Scion, University of Waikato / Publicly available
Dataset type:	Experimental
Data storage:	Data is currently held in an Excel spreadsheet by Haydon Jones.
Broad Land use / Landform sampled:	Exotic plantation forestry (first and yearly second rotation) / ridges, slopes, and gully floors in hill country.
Number of sites:	Samples taken from two adjacent 5 ha plots (pre and post-harvested) within the southern part of Mahurangi Forest.
Geographical spread:	Experimental site was located just south of Warkworth, northern Auckland.
Soil orders:	Ultic Soils (88 and 92 % of observations in pre and post-harvested plots respectively), Gley Soils, Recent Soils, Raw Soils.
Depth of sampling: 0-30 cm	No
Specific sampling depths:	Mineral soil 0-10 cm only.
Bulk Density measurements:	Yes (but measured on a 1.5-3.5 cm depth range at all sample points using the small-core developed by Peter Singleton – calculation of soil C stocks was not an original purpose of this sampling).
Multiple sampling through time:	No
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	GIS surfaces giving the locations of the plots and every individual sample point are available.
Soil description:	Yes (observed at every individual sample point in addition to full descriptions of representative profiles).
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	Yes
Land use history recorded:	Yes
If Yes above, then history for what period?	25+ yrs preceding last sampling (land use since sampling has been, or soon will be, changed to lifestyle blocks). Prior to the establishment of the forest, the land use was mainly sheep and beef grazing.
Management history available:	
History timeframe:	From about 30 years prior to sampling.
Management factors held:	Limited information regarding scrub clearance and forest management practices, in addition to a description of forest harvesting technique used,

was detailed in Jones (2004). However, no information on management practices while under agriculture was available.

Data held in what form:

Limited information is held in both electronic (Word) and hard-copy (thesis) form.

Mitigation opportunities to minimise or improve soil C:

Findings based on this dataset suggested that opportunities for soil C loss mitigation may arise through the retention of harvest residues on-site and by allowing the development of a general cover of grassy weeds during the establishment of the next rotation.

Supplementary information on soils:

Several other soil properties were measured in addition to total C: pH, macroporosity, and Bray available P, K, and Mg.

Associated publications:

Jones, H.S., 2004. Impacts of forest harvesting on the performance of soil-landscape modelling in a radiata pine forest, northern New Zealand. Unpublished PhD. thesis, lodged in the Library, University of Waikato, Hamilton.

Free-form database description

Origin:

This dataset was collected as part of a study investigating the effects of harvesting disturbance on the performance of a range of soil spatial prediction techniques at a first-rotation plantation forest site.

Collection methodology:

A full description of the site, trial design, and methodology used is given in Jones (2004).

Samples for chemical analysis were collected from the top 10 cm of mineral soil on a 16.5 m regular grid within two adjacent 5 ha plots (one harvested, the other not) using a small-diameter sampling tube. A total of 208 samples were collected within each plot. Samples for bulk density (and macroporosity) analysis were also collected on the same grid system using a small-core method (Drewry et al., 2002).

Sample analysis:

Total C was measured using a high-frequency induction furnace (Shimadzu solid sample module attached to the infrared detector of the Shimadzu 5000a TOC instrument). The total C content of the soil samples was determined by measuring the CO₂ produced using an infrared detector during the combustion (total combustion at 900° C) of 0.2 g of finely ground soil in an O₂ atmosphere.

Strengths and limitations:

Plots were established within areas that were managed using standard forest management practices for the site type. Estimated plot means were based on a large number of spatially distributed and geo-referenced samples giving sound data on which to compare plots. However, the main limitation is that the paired plots were not replicated. It is possible,

although unlikely, that the soil properties in the two plots were naturally different prior to the harvesting of one of them. The dataset is also limited by pertaining to one site only.

Assumptions and uncertainties:

It was assumed that the soil properties of both plots were similar prior to the application of the harvesting treatment. However, the reliance on this assumption gives-rise to uncertainty regarding the actual cause of the difference in C stocks between the plots. Nevertheless, the findings were consistent with other studies; giving weight to the validity of the assumption.

Main findings relevant to change in soil C stocks:

The main finding of the work in relation to soil C stocks was that hauler-based forest harvesting involving some soil compaction and, followed by residue retention and preservation of a grassy vegetation cover, may lead to an increase (of about 10 Mg ha⁻¹) in soil C stocks in the top 10 cm of mineral soil.

Future plans:

None, other than to publish the various findings of the wider study.

1.2.5 Puruki (Forest Harvesting Effects)

SUMMARY

Puruki forest consists of three sub-catchments planted in radiata pine which were harvested at age 24. Two catchments were harvested using a ground based method and the third by hauler logging. Soil samples in each catchment were collected before and after logging to compare the effect of the different logging methods on soil C. Results are published and showed hauler logging had little effect on soil C in the 0-0.1 m layer, while ground based logging reduced soil C by about 5 Mg ha⁻¹. However deeper soil sampling failed to show a significant difference between logging methods.

DATABASE DESCRIPTION

Name of the dataset:	Puruki Forest (Harvesting Effects).
Primary contact:	Dr Peter Beets, Scion
Data ownership / accessibility:	Scion, FRST, MFE / publicly available
Dataset type:	Experimental
Data storage:	Data is currently held in an Excel spreadsheet on a network drive accessible by Carbon Team staff.
Broad Land use / Landform sampled:	Exotic plantation forestry (<i>P. radiata</i>) / gently rolling to moderately steep slopes
Number of sites:	Three sub-catchments within the Puruki site were sampled.
Geographical spread:	The Puruki catchment is situated within the Paeroa Range, about 30 km south of Rotorua in the central North Island.
Soil orders:	Pumice Soils.
Depth of sampling: 0-30 cm	No
Specific sampling depths:	Mineral soil 0-10, 0-20, 20-50, 50-200, 0-100 cm (with different sample numbers collected from the various depth increments).
Bulk Density measurements:	Yes (at least 2 per plot)
Multiple sampling through time:	Yes
Pre 1990 soil sampling:	No
Site Location Information available:	Yes/No
Information Type:	GPS coordinates, and GIS maps
Soil description:	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	Yes
Land use history recorded:	Yes
If Yes above, then history for what period?	33+ yrs preceding last sampling and up to present. (Surface soils – 0-10 cm – were last sampled in 2006 to track changes through time).
Management history available:	
History timeframe:	For about the last 50 years.
Management factors held:	All silvicultural practices followed, harvesting techniques applied, and trial maintenance works.

Data held in what form: Both electronic and hard-copy records.

Mitigation opportunities to minimise or improve soil C:

Findings based on this dataset suggested that opportunities for soil C loss mitigation may arise through the selection of forest harvesting techniques that cause less soil physical disturbance than conventional ground-based harvesting (e.g. hauler-based harvesting).

Supplementary information on soils:

Total N %, and other soil properties are available.

Associated publications:

Beets, P.N. and Brownlie, R.K., 1987. Puruki experimental catchment: site, climate, forest management, and research. *New Zealand Journal of Forestry Science*, 17(2-3), 137-160.

Beets, P.N., Oliver, G.R., and Clinton, P.W., 2002. Soil carbon protection in podocarp/hardwood forest, and effects of conversion to pasture and exotic pine forest. *Environmental Pollution*, 116(S1): S63-S73.

Oliver, G.R., Beets, P.N., Garrett, L.G., Pearce, S.H., Kimberley, M.O., Ford-Robertson, J.B., and Robertson, K.A., 2004. Variation in soil carbon in pine plantations and implications for monitoring soil carbon stocks in relation to land-use change and forest site management in New Zealand. *Forest Ecology and Management*, 203(1-3): 283-295.

Free-form database description

Origin:

This dataset was collected primarily to determine and compare the impacts of hauler- and ground-based harvesting on soil C stocks. However, there was also an interest in establishing the intensity of sampling required to detect treatment differences in the variable C data.

Collection methodology:

Refer to Oliver et al. (2004) for a full description of the site, trial design, and methodology used in the study.

Three sub-catchments at Puruki were sampled – two (Rua and Toru) were harvested using a ground-based technique whereas the other (Tahi) was harvested using a hauler logging system. Thirty small (0.04 ha) permanent sample plots located across the site were sampled before harvest and again about 3 year after harvest. The 0-10 cm mineral soil depth range was sampled using small-diameter sampling tubes. Ten cores per plot were collected in a regular grid pattern prior to harvesting whereas 30 cores were collected per plot after harvesting. Cores were bulked by plot. For the assessment of bulk density, two 60 mm-diameter core samples were collected per plot prior to harvesting and ten cores per plot collected after harvesting.

Sample analysis:

Total C was measured on air-dried soil sieved to < 2mm using a LECO 2000 thermal combustion furnace. Fine-earth bulk density was measured on the volumetric core samples oven-dried at 105 °C.

Strengths and limitations:

A key strength of the study is that C stocks were measured in the same areas before and after harvesting so that any observed differences can unequivocally be attributed to harvesting disturbance. The primary limitation with this dataset is that it relates to only one set of soil and site conditions. However, the results are likely to be generally applicable to parts of the volcanic terranes in the central North Island. Data on C stock changes below 10 cm depth are limited by low sample numbers.

Assumptions and uncertainties:

It was assumed that both harvesting techniques would cause similar effects in all three sub-catchments to those actually observed.

Main findings relevant to change in soil C stocks:

The main findings of the study were that harvesting disturbance overall resulted in about a 3 Mg ha⁻¹ loss of soil C from the 0-10 cm depth range and that ground-based harvesting in particular was responsible for the loss — about 5 Mg ha⁻¹ was lost from the top 10 cm of mineral soil after ground-based harvesting in the Toru sub-catchment (Figure 1.6).

Future plans:

Continue regular sampling to track changes in the C stocks of the top 10 cm of mineral soil.

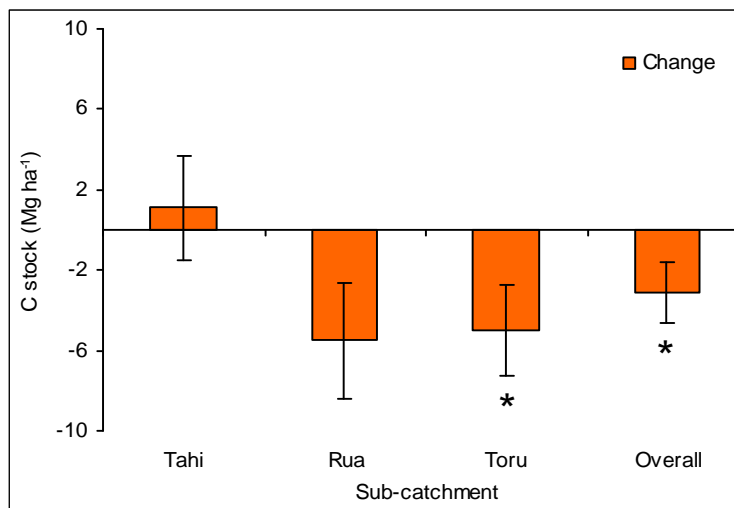


Figure 1.6. Change in soil C stocks (0-10 cm) following hauler- (Tahi) and ground-based (Rua and Toru) harvesting at Puruki. Overall change (across both harvesting treatments) is also given (after Oliver et al., 2004). Error bars represent the standard error.

1.2.6 LTSP Series I Trials (Residue Management and Fertiliser Effects)

SUMMARY

This experimental dataset has been collected to determine the impacts of different levels of harvest residue retention and site preparation on major soil types on short- and long-term site productivity and to provide information for management recommendations that will sustain productivity in exotic plantation forests of *Pinus radiata*. The tree biomass components (at harvest), forest floor materials and mineral soils (0-20cm) were sampled, and ecosystem mass and nutrient pools were quantified. The data is held by Scion in an Excel spreadsheet. Removal of both harvest residues and forest floor horizons resulted in significantly lower nutrient stocks at some sites (e.g. N at Woodhill; Mg at Kinleith).

DATABASE DESCRIPTION

Name of the dataset:	LTSP (Long-Term Site Productivity) Series I Trials (Residue Management and Fertiliser Effects).
Primary contact:	Dr Peter Clinton, Scion
Data ownership / accessibility:	Scion, FRST / publicly available.
Dataset type:	Experimental
Data storage:	Data is currently held in Excel spreadsheets by Peter Clinton.
Broad Land use/Landform sampled:	Second-rotation exotic plantation forestry (<i>P. radiata</i>) / Specific landform varies among trial sites – some are generally flat (e.g. Tarawera) sites whereas others are undulating or sloping (e.g. Golden Downs).
Number of sites:	Originally six trial sites (recently the number has reduced to four due to land use change — deforestation).
Geographical spread:	Includes six forest sites (three North Island and three South Island) from northwest Auckland (Woodhill) in the north to Otago (Berwick) in the south.
Soil orders:	Recent Soils (two sites), Pumice Soils, Brown Soils (two sites), and Pallic Soils.
Depth of sampling: 0-30 cm	No
Specific sampling depths:	Mineral soil 0-5 cm; 0-10, 10-20cm (each main plot); Soil samples to the depth of 60 or 100cm were also sampled from different sites pre-planting.
Bulk Density measurements:	Yes
Multiple sampling through time:	Yes
Pre 1990 soil sampling:	Yes
Site Location Information available:	Yes
Information Type:	Map coordinates (grid references).
Soil description:	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	Yes
Land use history recorded:	Yes

If Yes above, then history for what period? Varies with trial site but up to 57+ yrs preceding last sampling.

Management history available:

History timeframe:

Varies with trial site but up to about 50 years ago until present.

Management factors held:

Specific information held on harvest residue management treatments (core treatments are: whole-tree harvest, stem-only harvest, and whole-tree harvest + forest floor removal) and fertilisers rates applied, and on other silvicultural manipulations (e.g. weed control, pruning, and thinning).

Data held in what form:

Ancillary data regarding the trial is variously held in both electronic form on a network drive and in trial notebooks (held and maintained by Graeme Oliver, Doug Graham, Stephen Pearce, Peter Clinton and Dave Henley).

Mitigation opportunities to minimise or improve soil C:

Findings based on this dataset suggested that there are opportunities for soil C loss mitigation via the retention of both harvest residues and forest floor materials on-site.

Supplementary information on soils:

Total N (% and stocks) data is also included in this particular dataset. A range of tree growth, above-ground biomass, woody decay, and soil property data also exists in various other datasets relating to these trials.

Associated publications:

Lowe, A.T. 1994: Forest-floor and below-ground nutrient concentrations and statistical summaries after 5-years at Woodhill (AK1029). NZ FRI Project Record No. 4250

Smith, C.T.; Lowe, A.T.; Skinner, M.F.; Beets, P. N.; Schoenholtz, S.H.; Fang, Shengzuo 2000 Response of radiata pine forests to residue management and fertilisation across a fertility gradient in New Zealand. *Forest Ecology and Management*. 138, 203-223.

Smaill J.S., Clinton P.W., Greenfield, L.G. 2008 Postharvest organic matter removal effect on FH layer and mineral soil characteristics in four New Zealand *Pinus radiata* plantations. *Forest Ecology and Management*. (in press)

Smaill, S.; Clinton, P.W.; Greenfield, L.G. 2008. Variations in litter fall, FH layer and mineral soil characteristics in response to nitrogenous fertilisation across six New Zealand *Pinus radiata* plantations. *Forest Ecology and Management*. (in press)

Free-form database description

Origin:

This dataset was collected primarily to investigate the impacts of three different harvest residue management treatments (whole-tree harvest, stem-only harvest, and whole-tree harvest + forest floor removal) on ecosystem biomass and nutrient pools on major soil types in NZ plantation forests, to determine the relationships between soil physical and chemical variables and forest productivity, and relationships between these same variables and

management practices. At Burnham, effect of different harvesting intensity on wood quality issues was also investigated.

Collection methodology:

A full description of the site, trial design, and methodology used is given in Lowe (1994), Smith et al (2000) and Smaill et al. (2008, in press).

Sample analysis:

For a detailed description see Smaill et al. (2008). Mineral soil and forest floor samples were air-dried before being analysed for total C and N using a LECO Corporation CNS-2000 Elemental Analyser.

Strengths and limitations:

Strengths – All 6 trials were blocked split-plot designs. All trials contained four blocks (replications) except Woodhill, which contained only three blocks. Pre-harvest biomass data, and forest floor, soil and understorey data are available.

Limitation – Soil chemistry data from the top 20cm was obtained for each main plot at Berwick and Burnham, and at Kinleith and Tarawera (for core treatments FF, WT and LO only). At Golden Downs and Woodhill, only site means were obtained. In addition, soil core samples were only collected to a depth of 2.5cm in a recent study (Smaill et al., 2008). Soil samples collected pre-planting are not plot-based.

Assumptions and uncertainties:

Uncertainty of future funding provision

Main findings relevant to change in soil C stocks:

Increasing organic matter removal significantly decreased the mass of FH layer in the treatment plots, the concentration of carbon in the FH layer and mineral soil, the pool of carbon stored in the FH layer (Smaill et al., 2008).

Future plans:

Soil samples to the depth of 0-20cm are archived from all six LPST I trials and could be analysed for total soil C if funding is available. A Scion post-doc will be recruited to do research related soil C in LTSP I trials.

1.2.7 LTSP Series II Trials (Compaction, Fertiliser, and Species Effects)

SUMMARY

This trial series was established to determine the key productivity drivers for two contrasting species – *Pinus radiata* and *Cupressus lusitanica*. Densely planted mini-plots of each species were planted at 35 sites representing the major soil and climate domains for plantation forestry across NZ. Fertiliser (with/without) and disturbance (with/ without) were included as treatments. The design was a replicate factorial (species x fertiliser x disturbance) with a single replicate of eight plots at each site. The plots were harvested at age 4. At each site a single large permanent sample plot has been established which has been split to give two vegetation control treatments. The latter plots will be carried through to maturity and used to confirm (or otherwise) the results obtained from the mini-plots.

DATABASE DESCRIPTION

Name of the dataset:	LTSP Series II ‘Site Quality’ Trials (Compaction, Fertiliser, and Species Effects).
Primary contact:	Dr Peter Clinton, Scion
Data ownership / accessibility:	Scion, FRST / publicly available
Dataset type:	Experimental
Data storage:	Data is currently held in an Excel spreadsheet on a network drive accessible by Carbon Team staff.
Broad Land use/Landform sampled:	Second/third-rotation exotic plantation forestry (<i>P. radiata</i> and <i>C. lusitanica</i>) / Specific landform varies among trial sites – some are generally flat (e.g. Balmoral) sites whereas others are undulating or sloping (e.g. Aniseed Valley).
Number of sites:	35 sites. Each site has 2 types of plots (envelope plots at 3 by 3 m, and new PSPs at 40 x 40 m). Vegetation gradient plots (40 by 40 m) were installed at 9 of 35 sites.
Geographical spread:	Includes 35 forest sites (18 North Island and 17 South Island) from Northland to Southland.
Soil orders:	Brown soils (13 sites), Allophanic soils (5 sites), Pumice soils (3 sites), Pallic soils (3 sites), Recent soils (3 sites), Ultic soil (3 sites), Raw soils (1 site), Podzol soils (4 sites)
Depth of sampling: 0-30 cm	No
Specific sampling depths:	Soil was sampled by horizons pre-planting, and sampled to the depth of 0-10cm at 4 years after establishment)
Bulk Density measurements:	Yes (From soil pit and around plots. 5 replicates per disturbance class)
Multiple sampling through time:	Yes
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	GPS coordinates for each trial site are available
Soil description:	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	No

Land use history recorded:	Yes
If Yes above, then history for what period?	38+ yrs preceding last sampling and up to present (mineral soils (0-10 cm) were last sampled in 2006 from PSPs to track changes through time).
Management history available:	Yes
History timeframe:	The trials were planted about 6-8 years ago and the previous land use was first/second-rotation exotic plantation forestry. We hold information on management history over the period of 6-8 years, but the land owners may hold information on the management history of the first/second rotation and perhaps the preceding land use.
Management factors held:	Specific information held on 2 x 2 x 2 (disturbance, fertilization and species) factorial design for Productivity Envelope plots, 4 weed control treatments (0, 50, 75 and 100%) for Vegetation Gradient plots, and 2 weed control treatments (0 and 100%) for New PSPs.
Data held in what form:	Ancillary data regarding the trial is variously held in both electronic form on a network drive or in trial notebooks (held and maintained by Graham Coker).

Mitigation opportunities to minimise or improve soil C:

Effects of disturbance (mainly compaction), fertiliser and species on soil C in 0-0.1 m layer can be determined from envelope plots, effects of vegetation management can be determined from psp plots at each site.

Supplementary information on soils:

Total N (%), extractable NO₃-N and NH₄-N, Mineralisable-N, Olsen P, Bray P, total P, organic P, soluble P, exchangeable cations, CEC, base saturation (%) and pH data are also included in this particular dataset. A range of tree growth, above- and below-ground biomass and allocation, and soil physical property data are also available.

Associated publications:

- Davis, M.R., Coker, G., Parfitt, R.L., Simcock, R., Clinton, P.W., Garrett, L.G., Watt, M.S. (2007) Relationships between soil and foliar nutrients in young densely planted stands of *Pinus radiata* and *Cupressus lusitanica*. *Forest Ecology and Management* 240, 122-130.
- Kiyvyra, A.L., Schoenholtz, S.H., Clinton, P.W., Coker, G.W.R., Burger, J.A., Watt, M.S., Graham, D. (2007) Fertiliser and site disturbance effects on nitrogen mineralization, litter decomposition, wood decay, and daily soil water for planted *Pinus radiata* and *Cupressus lusitanica* forests across an environmental gradient in New Zealand. In: *Proceedings of the international symposium on Forest Soils and Ecosystem Health*, Noosa, Australia. August 19th -23rd 2007. pp. 65-66.
- Watt, M.S., Coker, G., Clinton, P.W., Davis, M., Parfitt, R., Simcock, R., Garrett, L., Payn, T., Richardson, B., Dunningham, A. (2005). *Defining sustainability of plantation*

forests through identification of site quality indicators influencing productivity- a national view for New Zealand. *Forest Ecology and Management* 216, 51-63.

Watt, M.S., Davis, M.R., Clinton, P.W., Coker, G., Ross, C., Dando, J., Parfitt, R.L., Simcock, R. 2008 Identification of key soil indicators influencing plantation productivity and sustainability across a national trial series in New Zealand. *Forest Ecology and Management*. (in press)

Free-form database description

Origin:

See above publications

Collection methodology:

Envelope plots – Initial soil samples (0-0.1 m) were collected at 5 points in each of the four undisturbed plots to give 20 cores per bag. This process was repeated for the four disturbed plots at each location. Sampling points were at least 1 m apart near the centre of the plots. These samples were bulked by disturbance class. Further samples (also 0-0.1 m) were collected after four years when plots were harvested. Sixteen samples per plot were collected on a grid pattern with sample points located equidistant between trees. Samples were bulked by plot.

Sample analysis:

Mineral soil samples were air-dried before being analysed for total C and N using a LECO Corporation CNS-2000 Elemental Analyser.

Strengths and limitations:

Strengths – A four-tiered approach to productivity monitoring is used. This comprises a hierarchy of three plot types collectively called the "Site Quality Plots", with all three types installed at major soil order locations for New Zealand plantation forests (i.e. Productivity Envelope plots, Vegetation Gradient plots, and New PSPs). Beside soil related data, water balance model, soil respiration, photosynthesis, and litter and wood decay rate data are also available.

Limitations – Currently, only New PSPs are maintained for long-term productivity data, and soil was sampled at 0-10cm.

Assumptions and uncertainties:

Uncertainty of future funding.

Main findings relevant to change in soil C stocks:

No information available.

Future plans:

Soil samples (0-10cm) collected during the rotation are archived from all 35 LPST II trials and could be analysed for total soil C if funding is available.

1.2.8 LTSP Series III Trials (Genotype and Species Effects)

SUMMARY

This trial series has been established at 14 sites that are expected to exhibit key nutrient deficiencies of radiata pine. Moisture availability is also limiting at some sites. The trials have been planted with different genotypes of *Pinus radiata* and *Cupressus* species, and of *Pseudotsuga menziesii* at two locations. The aim of the trials is to examine the genetic range of variability of response to limiting nutrient and moisture availability.

DATABASE DESCRIPTION

Name of the dataset:	LTSP Series III 'Genetics by Environment' Trials.
Primary contact:	Dr Jianming Xue, Scion
Data ownership / accessibility:	Scion, FRST / publicly available
Dataset type:	Experimental / Inventory
Data storage:	Data is currently held in an Excel spreadsheet on a network drive accessible by Carbon Team staff.
Broad Land use/Landform sampled:	Second-rotation exotic plantation forestry (<i>P. radiata</i>) / Specific landform varies among trial sites – some are generally flat (e.g. Balmoral) sites whereas others are undulating or sloping (e.g. Aniseed Valley).
Number of sites:	14 sites comprising four radiata pine plots. Four plots of cypress and/or Douglas-fir also included in some sites.
Geographical spread:	14 sites nationwide (6 North Island and 8 South Island) from northwest Auckland (Woodhill Forest) in the north to Otago (Lawrence) in the south.
Soil orders:	Brown soils (5 sites), Pallic soils (2 sites), Pumice soils (3 sites), Podzol soils (2 sites), Recent soils (2 sites)
Depth of sampling: 0-30 cm	No
Specific sampling depths:	Soil was sampled at the depths of 0-10 and 10-20cm pre-planting,
Bulk Density measurements:	No
Multiple sampling through time:	No
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	GPS coordinates
Soil description:	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	No
Land use history recorded:	Yes
If Yes above, then history for what period?	36+ yrs preceding last sampling & up to present.
Management history available:	Yes
History timeframe:	Trials were planted about 4-6 years ago and the previous land use was first/second-rotation exotic

Management factors held:	plantation forestry. We hold information on management history over the period of 4-6 years but the land owners may hold information on the management history of the first/second rotation and perhaps the preceding land use.
Data held in what form:	Site preparation varied with sites. Soils were only ripped in some sites, but all sites were spot-sprayed pre-planting. Trial maintenance activities, e.g. weed control, fertilization recorded Limited management history data is held in electronic trial information sheets and other files on a network drive.

Mitigation opportunities to minimise or improve soil C:

Nil at present, will provide information on tree species effects on soil C.

Supplementary information on soils:

Soil total N (%) and pH are also included in this particular dataset. A range of tree growth data also exists in various other datasets relating to these trials.

Associated publications:

Xue, J., Clinton, P.W., Davis, M., Siddiqui, T. Beets, P. Leckie, A. Graham, D.J. Genotypic variation of foliar nutrient concentrations and $\delta^{15}\text{N}$ in relation to tree growth of 5-year-old *Pinus radiata* at two contrasting sites in New Zealand. Submitted to *Forest Ecology and Management*.

Olykan, S.T., Xue, J.M., Clinton, P. W., Skinner, M.F., Graham, D.J., Leckie, A.C. Effect of boron fertiliser, weed control and genotype on foliar nutrients and tree growth of juvenile *Pinus radiata* at two contrasting sites in New Zealand, *Forest Ecology and Management*, 255: 1196–1209

Free-form database description**Origin:**

Above publications.

Collection methodology:

Soil samples were collected between tree rows (25 cores/plot) from each plot at depths of 0-10 and 10-20 cm.

Sample analysis:

Mineral soil samples were air-dried before being analysed for total C and N using a LECO Corporation CNS-2000 Elemental Analyser.

Strengths and limitations:

Strengths – All 14 trials were a randomised complete block (plot) design with 4 replications (plots). These trials cover a range of site conditions, soil fertility and water availability. Farm sites have also been included in some areas to offer contrasts in nutrient availability. Where possible, sites have been established in areas where a Site Quality plot is already located.
Limitation – up to date, soil samples (0-20cm) were only collected from all trials at planting.

Assumptions and uncertainties:

Uncertainty of future funding.

Main findings relevant to change in soil C stocks:

No information available.

Future plans:

Soil samples (0-10cm) collected at planting are archived from all 14 LPST III trials and could be analysed for total soil C if funding is available.

1.2.9 Tikitere Agroforestry (Stocking Effects)

SUMMARY

This trial has been completed. Various studies have been undertaken throughout the life of the plantation that included measurements of soil C. The trial allowed comparison of the effect of tree stocking (density) on soil C. Results are contained in published papers.

DATABASE DESCRIPTION

Name of the dataset:	Tikitere Agroforestry (Stocking Effects).
Primary contact:	A. Ghani (AgResearch) and S. Saggar (Landcare Research).
Data ownership / accessibility:	Agresearch / published results publicly available.
Dataset type:	Experimental
Data storage:	Uncertain.
Broad Land use/Landform sampled:	Uncertain.
Number of sites:	1
Geographical spread:	Represents one site only (Tiketere Agroforestry Research Area near Rotorua).
Soil orders:	Typic Orthic Pumice Soils.
Depth of sampling: 0-30 cm	Yes (in part)
Specific sampling depths:	Perrot et al. (1999): primarily 0-7.5, but also 0-3, 3-7.5, and 7.5-15 cm; Saggar et al. (2001): 0-10 and 10-20 cm (for C concentration); Scott et al. (2006): 0-10, 10-20, and 20-50 cm.
Bulk Density measurements:	Yes (some)
Multiple sampling through time:	Yes
Pre 1990 soil sampling:	Yes
Site Location Information available:	Yes (probably)
Information Type:	Probably just map coordinates
Soil description:	Yes (probably)
Are soil samples archived?	Uncertain.
Total change in C or rate of change in soil C recorded:	Yes
Land use history recorded:	Yes
If Yes above, then history for what period?	Since at least 1973 when the trial was established.
Management history available:	
History timeframe:	Uncertain.
Management factors held:	Uncertain.
Data held in what form:	Uncertain.
Mitigation opportunities to minimise or improve soil C:	
	At stocking rates above 200 stems per ha there is little potential for changes in soil C stocks to occur with changes to stocking.
Supplementary information on soils:	
	Total P, organic P, inorganic P, CEC, EC, microbial C, basal respiration, microbial P, and P retention.
Associated publications:	

- Perrott, K.W., Ghani, A., O'Connor, M.B., Waller, J.E., and Hawke, M.F., 1999. Tree stocking effects on soil chemical and microbial properties at the Tikitere agroforestry research area. *New Zealand Journal of Forestry Science*, 29(1): 116-130.
- Saggar, S., Hedley, C.B., and Salt, G.J., 2001. Soil microbial biomass, metabolic quotient, and carbon and nitrogen mineralisation in 25-year-old *Pinus radiata* agroforestry regimes. *Australian Journal of Soil Research*, 39(3): 491-504.
- Scott, N.A., Tate, K.R., Ross, D.J., and Parshotam, A., 2006. Processes influencing soil carbon storage following afforestation of pasture with *Pinus radiata* at different stocking densities in New Zealand. *Australian Journal of Soil Research*, 44(2): 85-96.

Free-form database description

Origin:

Studies undertaken to assess the effects of agroforestry plantings on soils and pasture production.

Collection methodology:

Refer to the above publications.

Sample analysis:

Refer to the above publications.

Strengths and limitations:

Refer to the above publications.

Assumptions and uncertainties:

Refer to the above publications.

Main findings relevant to change in soil C stocks:

Refer to discussion of above studies in the review of NZ literature given earlier in this report.

Future plans:

Unknown.

1.2.10 Kyoto Forest Inventory (Age Class Effects)

SUMMARY

This inventory dataset is currently under development, but when complete will provide an insight into the differences in soil C concentration in the 0-5 cm depth range under different age classes of *Pinus radiata* and some other common plantation species (a proxy for change in soil C over the course of a rotation).

DATABASE DESCRIPTION

Name of the dataset:	Kyoto Forest Inventory Plots (Forest C Inventory).
Primary contact:	Dr Thomas Paul, Scion.
Data ownership / accessibility:	MFE, MAF / Publicly available in future
Dataset type:	Inventory
Data storage:	Data is currently held by MFE as a part of the LUCAS system and administered by Interpine.
Broad Land use/Landform sampled:	First rotation exotic plantation forestry / flat to hilly land.
Number of sites:	42 sites have been sampled to date but will ultimately include about 250 sites.
Geographical spread:	Plots sampled to date are in the North Island and Nelson/Marlborough region but ultimately national coverage will be achieved.
Soil orders:	Currently unknown – not profiles not being described on-site – but could be identified using soil class maps.
Depth of sampling: 0-30 cm	No
Specific sampling depths:	Litter and 0-5 cm mineral soil
Bulk Density measurements:	No
Multiple sampling through time:	No (but ultimately will)
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	GIS, GPS coordinates
Soil description:	No
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	No
Land use history recorded:	No (presumably not forest)
If Yes above, then history for what period?	For up to 18 yrs preceding sampling
Management history available:	
History timeframe:	N/A
Management factors held:	Age class
Data held in what form:	In Field-master database
Mitigation opportunities to minimise or improve soil C:	
	N/A

Supplementary information on soils:

Total N (%) data are also included in this dataset.

Associated publications:

None at this stage.

Free-form database description**Origin:**

This dataset is being developed as part of the wider Kyoto (post 1989) forest biomass C stock inventory undertaken to support and validate forest C stock estimates made using LiDAR.

Collection methodology:

Mineral soil samples are collected using a small-diameter tube sampler. 20 cores are collected from within the central plot and bulked.

Sample analysis:

Samples were air-dried, and passed-through a 2 mm sieve, before being analysed for total C and N using a LECO FPS-2000 CNS thermal combustion furnace.

Strengths and limitations:

Ultimately, the strengths of this dataset will include the national coverage of sites across a range of soil and climatic conditions and a measure of soil C under operational forest management conditions. However, this dataset is limited by not sampling to 30 cm depth and the absence of bulk density data (although a pedotransfer function approach could potentially be used to estimate bulk density).

Assumptions and uncertainties:

N/A

Main findings relevant to change in soil C stocks:

Data collection and analysis is yet to be completed.

Future plans:

Sites will be re-sampled and remeasured in 2012.

1.2.11 Balmoral-Waitaki (Stocking Effects)

SUMMARY

This trial was established in 1993 as an agroforestry trial in a sub-humid zone in the Mackenzie Basin. It contains 4 replicates of 4 tree spacings (0, 250, 500, 750 sph) of *Pinus nigra* planted into undeveloped grassland. Half the trial has been sown with clovers and fertilised. Soil C was measured (0-0.3 m) between tree rows at year 5. As rows were 4 m apart and tree growth is slow in the environment this measurement was considered equivalent to time zero. Soil C was also measured at year 10 (in 2003). No change in soil C was evident. Above ground forest biomass was also measured at year 10. Results have been published.

DATABASE DESCRIPTION

Name of the dataset:	Balmoral-Waitaki (Stocking Effects)
Primary contact:	Murray Davis, Scion
Data ownership / accessibility:	Scion, Publicly available
Dataset type:	Experimental
Data storage:	Excel spreadsheet, Murray Davis
Broad Land use/Landform sampled:	Unimproved grassland and afforested unimproved grassland/High country terrace
Number of sites:	One
Geographical spread:	Trial site is located on Balmoral Station, South of Lake Tekapo, Mackenzie Basin
Soil orders:	Orthic Brown
Depth of sampling: 0-30 cm	Yes
Specific sampling depths:	0-10, 10-20, 20-30
Bulk Density measurements:	Yes
Multiple sampling through time:	Yes
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	Map coordinates
Soil description:	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	Yes
Land use history recorded:	Yes
If Yes above, then history for what period?	Approximately 100 yrs
Management history available:	
History timeframe:	Year-by-year data for 10 yrs proceeding last sampling, when the trial was planted, prior to that anecdotal history of grazing of unimproved grassland for previous 100 years
Management factors held:	Tree crop type, tree stocking rate, fertiliser type/rate and legume species introduction (for part of trial)
Data held in what form:	Excel spreadsheet

Mitigation opportunities to minimise or improve soil C:

Soil C has not changed in first 10 years since afforestation, is expected to increase as forest floor develops. Re-measurement might indicate that soil C could be increased through nitrogen input from legumes and fertiliser application.

Supplementary information on soils:

% total N for all plots, site data for pH, phosphorus (total, inorganic, organic, Bray-2, Olsen) and exchangeable cations.

Associated publications:

Davis M, Nordmeyer A, Henley D, Watt M. 2007. Ecosystem carbon accretion ten years after afforestation of depleted sub-humid grassland planted with three densities of *Pinus nigra*. *Global Change Biology* 13, 1414-1422.

Free-form database description**Origin:**

Above publication

Collection methodology:

Plot soil coring, initial sampling between tree rows (10 cores/plot), at year 10 random sampling along transects (25 cores/plot)

Sample analysis:

C and N analysis of < 2 mm mineral soil and pine and herbaceous roots in cores by LECO analyser

Strengths and limitations:

Strengths – replicated trial (4 reps), relatively uniform flat site, different tree stockings including unplanted control plots, unimproved and improved (fertiliser + legume) treatments, has tree productivity, biomass and carbon information, control over trial through joint venture ownership with land owner.

Limitations – Tree species is *Pinus nigra* rather than *Pinus radiata*

Assumptions and uncertainties:

Uncertainty of future funding provision

Main findings relevant to change in soil C stocks:

Afforestation at any tree stocking had no effect on soil C stocks at age 10, as expected for a low productivity semi-arid site

Future plans:

Trial re-measurement at 5-year intervals, next measurement 2008

1.2.12 Orton Bradley (Species Effects)

SUMMARY

This trial was planted in pasture in 1999 at Orton Bradley Park on Banks Peninsula to compare the impact of 3 tree species (*Pinus radiata*, *Eucalyptus nitens*, *Cupressus macrocarpa*) on soil properties. The trial has 4 replicates. Soil was sampled (0-0.3 m) in 1999 and 2004 and is due for re-sampling at year 10 in 2004.

DATABASE DESCRIPTION

Name of the dataset:	Orton Bradley (Species Effects)
Primary contact:	Dr Leo Condon, Lincoln University
Data ownership / accessibility:	Dr Leo Condon, Lincoln University
Dataset type:	Experimental
Data storage:	Excel spreadsheets, L. Condon
Broad Land use/Landform sampled:	Pasture afforested in 1999 with <i>Pinus radiata</i> , <i>Cupressus macrocarpa</i> , and <i>Eucalyptus nitens</i> / Hill country slopes
Number of sites:	One
Geographical spread:	Located at Orton Bradley Park, Banks Peninsula
Soil orders:	Pallic
Depth of sampling: 0-30 cm	Yes
Specific sampling depths:	0-5, 5-10, 10-20, 20-30 cm depths
Bulk Density measurements:	Yes
Multiple sampling through time:	Yes
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	Map coordinates
Soil description:	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	Yes
Land use history recorded:	Yes
If Yes above, then history for what period?	For 25+ yrs proceeding last sampling
Management history available:	
History timeframe:	Year-by-year data for 5 yrs since planting, anecdotal history of pasture grazing by sheep prior to planting
Management factors held:	Tree crop type, tree stocking rate, thinning and pruning history, fertiliser type/rate (nil since planting)
Data held in what form:	doc files
Mitigation opportunities to minimise or improve soil C:	
	Soil C is expected to decline following afforestation, and be mitigated by some amount with development of a forest floor
Supplementary information on soils:	
	% total N, total S, total, organic, inorganic and Olsen P, CEC and exchangeable cations for all samples

Associated publications:

No

Free-form database description**Origin:**

File notes

Collection methodology:

Soil coring

Sample analysis:

Full chemical analysis on < 2 mm soil

Strengths and limitations:

Strengths – Replicated trial (3 reps) designed to evaluate and compare the effect of three tree species on soil properties. Chronosequence sampling, sampled at time of planting and 5 year intervals.

Assumptions and uncertainties:

Uncertainty of future funding provision

Main findings relevant to change in soil C stocks:

There has been a reduction in soil C mass at year 5 of 2-5 Mg ha⁻¹. No statistical analysis has been undertaken at this stage.

Future plans:

Continued sampling at 5 year intervals

1.2.13 Boron trial series (Herbicide Effects)

SUMMARY

Has potential to provide some data on the effects of weed removal on soil C stocks across New Zealand.

DATABASE DESCRIPTION

Name of the dataset:	Effect of boron on wood quality and the relationship between soil moisture, weed control and foliar boron
Primary contact:	Drs Peter Clinton/Jianming Xue, Scion
Data ownership / accessibility:	Scion, FFR / publicly available
Dataset type:	Experimental
Data storage:	Data is currently held in an Excel spreadsheet on a network drive accessible by Carbon Team staff.
Broad Land use/Landform sampled:	First to third rotation exotic plantation forestry (<i>P. radiata</i>) / Specific landform varies among trial sites – some are generally flat (e.g. Balmoral) sites whereas others are gently undulating or sloping (e.g. Lake Taupo).
Number of sites:	4 sites (FR358/1-4), each comprising 44 or 32 radiata pine plots.
Geographical spread:	4 sites nationwide (2 North Island and 2 South Island). FR358-1 is located in Balmoral Forest, North Canterbury, FR358-2 in Lake Taupo Forest, Taupo, FR358-3 in Tekapo, McKenzie Basin, FR358-4 in Tungrove Forest, Awarua.
Soil orders:	Brown Soils (3 sites, Balmoral, Tekapo and Tungrove); Pumice Soil (1 site, Lake Taupo)
Depth of sampling: 0-30 cm	No
Specific sampling depths:	Soil was sampled at the depths of 0-10 and 10-20cm from each plot of 4 trials, and at the depth of 20-50cm from the plots of B0, B8, & B32 at Lake Taupo and Tungrove trials.
Bulk Density measurements:	No
Multiple sampling through time:	Yes
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	GPS coordinates
Soil description:	No
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	No
Land use history recorded:	Yes
If Yes above, then history for what period?	30+ yrs preceding last sampling and up to present
Management history available:	
History timeframe:	The trials were planted between 1998 and 2002 and the previous land use was first/second-

- rotation exotic plantation forestry. We hold information on management history over the period of 6-10 years but the land owners may hold information on the management history of the first/second rotation and perhaps the preceding land use.
- Management factors held:** Site preparation varied with sites. Soils were ripped in all sites, and weeds spot-sprayed pre-planting. Various trial maintenance activities such as weed control, fertilization are recorded.
- Data held in what form:** Limited management history data is held in electronic trial information sheets and other files on a network drive.
- Mitigation opportunities to minimise or improve soil C:**
To be determined.
- Supplementary information on soils:**
Besides soil total C, soil total N (%), pH, Bray 2 P and Bray cations are also included in this particular dataset. A range of tree growth and foliar nutrient data also exist in various other datasets relating to these trials.
- Associated publications:**
- Olykan, S.T., Xue, J.M., Clinton, P. W., Skinner, M. F., Graham, J.D., Leckie, A. C. 2008 Effect of boron fertilizer, weed control and genotype on foliar nutrients and tree growth of juvenile *Pinus radiata* at two contrasting sites in New Zealand *Forest Ecology and Management* 255, 1196-1209.
- Xue, J.M., Clinton, P. W., Graham, J.D. and Leckie, A. C. 2006 Effects of boron fertilizer and weed control on soil available boron, foliar nutrients and tree growth of 6-year-old radiata pine at two contrasting sites. In: *Proceedings of New Zealand Soil Science Society Conference*, Rotorua, November 2006
- Xue, J.M., Clinton, P. W., Graham, J.D. and Leckie, A. C. 2006 Effects of boron fertilizer and weed control on soil available boron, foliar nutrition and tree growth of radiata pine grown at four sites. In: *Proceedings of 18th IUSS Congress*, Philadelphia, Pennsylvania, USA, July 2006
- Xue, J.M., Clinton, P. W., Payn, T., Graham, J.D. and Leckie, A. C. Effect of boron, genotype and site conditions on internal checking of radiata pine. In: *Proceedings of XXII IUFRO Conference*, Brisbane, Australia, August 2005.
- Xue, J.M., Clinton, P. W., Graham, J.D. and Leckie, A. C. 2006 Effect of Boron on Wood Quality in Balmoral trial (FR358-1) update 4, 2006.
- Xue, J.M., Clinton, P. W., Graham, J.D. and Leckie, A. C. 2006 Effect of Boron on Wood Quality in Lake Taupo trial (FR358-2) update 4, 2006.
- Xue, J.M., Clinton, P. W., Graham, J.D. and Leckie, A. C. 2006 Effect of Boron on Wood Quality in Tekapo trial (FR358-3) update 3, 2006.
- Xue, J.M., Clinton, P. W., Graham, J.D. and Leckie, A. C. 2006 Effect of Boron on Wood Quality in Tungrove trial (FR358-4) update 2, 2006.

Free-form database description**Origin:**

Above update report

Collection methodology:

Soil samples were collected from cultivated areas about half a metre from trees (around 30 cores/plot) in each plot at depths of 0-10 and 10-20 cm.

Sample analysis:

Mineral soil samples were air-dried before being analysed for total C and N using a LECO Corporation CNS-2000 Elemental Analyser.

Strengths and limitations:

Strengths – A blocked factorial designs of 4 or 5 boron rates \times 2 weed controls with 4 replications (blocks) was used for the trials. These trials cover a range of site conditions and soil fertility. One extra treatment (weed control plus balanced fertilizers) has also been included in 2 of these 4 trials for comparison.

Limitation – up to date, total soil C was only analysed from the control plot of these trials at establishment.

Assumptions and uncertainties:

Uncertainty of future funding provision

Main findings relevant to change in soil C stocks:

No information available

Future plans:

Soil samples (0-10, 10-20cm) could be analysed from each plot of all trials to determine the effect of fertilization and weed control on total soil C change if funding is available.

1.2.14 Mid-rotation N and P trials (Fertiliser Effects)

SUMMARY

Has potential to provide some additional data on the effects of fertiliser application on soil C stocks across New Zealand.

DATABASE DESCRIPTION

Name of the dataset:	Nitrogen and phosphorus responses in mid- to late rotation stands (FR467/1-5 trial series)
Primary contact:	Drs Peter Clinton/Jianming Xue, Scion
Data ownership / accessibility:	Scion, FFR / publicly available
Dataset type:	Experimental
Data storage:	Data is currently held in an Excel spreadsheet on a network drive accessible by Carbon Team staff.
Broad Land use/Landform sampled:	Second -rotation exotic plantation forestry (<i>P. radiata</i>) / Specific landform varies among trial sites – some are generally flat sites (e.g. Kaingaroa) whereas others are undulating or sloping (e.g. Golden Downs).
Number of sites:	5 sites, each comprising 12 or 14 radiata pine plots.
Geographical spread:	5 sites nationwide (3 NI and 2 SI). Two of the sites (FR467 1-2) are approximately 16 km and 30 km northwest of Dargaville, NI, the other two (FR467 3-4) about 35 km and 45 km southwest of Richmond, Nelson, SI. Another site (FR467-5) is located at Kaingaroa, central NI.
Soil orders:	Brown Soils (4 sites); Pumice Soil (1 site)
Depth of sampling: 0-30 cm	No
Specific sampling depths:	Soil was sampled at the depths of 0-10 and 10-20cm post-planting from each plot.
Bulk Density measurements:	No
Multiple sampling through time:	No
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	GPS coordinates
Soil description:	No
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	No
Land use history recorded:	Yes
If Yes above, then history for what period?	30+ yrs preceding last sampling and up to present
Management history available:	
History timeframe:	Trials were planted between 1982 and 1992 and the previous land use was first/second-rotation exotic plantation forestry. We hold information on management history over the period of 2-4 years but the land owners may hold information on the management history of the first/second rotation and perhaps the preceding land use.

Management factors held:

Site preparation varied with sites. Soils were only ripped in some sites, but all sites were spot-sprayed pre-planting. Various trial maintenance activities such as weed control, fertilization are recorded.

Data held in what form:

Limited management history data held in electronic trial information sheets and other files on a network drive.

Mitigation opportunities to minimise or improve soil C:

To be determined.

Supplementary information on soils:

Besides soil total C, soil total N (%), pH, Bray 2 P and Bray cations are also included in this particular dataset. A range of tree growth and foliar nutrient data also exist in various other datasets relating to these trials.

Associated publications: (in journals, proceedings, client reports to MFE, MAF, Regional Councils, etc.)

Xue, J., Graham, J.D., Clinton, P.W. 2006 Nitrogen and phosphorus responses in mid to late rotation stands (FR467) update 1.

Xue, J., Graham, J.D., Clinton, P.W. 2008 Nitrogen and phosphorus responses in mid to late rotation stands (FR467) update 2.

Free-form database description

Origin: Above update report

Collection methodology:

Soil samples were collected between tree rows (30 cores/plot) from each plot at depths of 0-10 and 10-20 cm.

Sample analysis:

Mineral soil samples were air-dried before being analysed for total C and N using a LECO Corporation CNS-2000 Elemental Analyser.

Strengths and limitations:

Strengths – A blocked factorial designs of 2 N × 2 P with 3 replications (blocks) was used for all 5 trials. These trials cover a range of site conditions and soil fertility. 2 extra treatments (weed control with or without fertilizer) have also been included in 2 of these 5 trials for comparison. *Limitation* – up to date, soil samples (0-10, 10-20cm) were only collected from each plot of 5 trials at establishment.

Assumptions and uncertainties:

Uncertainty of future funding provision

Main findings relevant to change in soil C stocks:

No information available

Future plans:

Soil samples (0-10, 10-20cm) could be re-collected from all trials next year to determine the effect of fertilization and weed control on total soil C change if funding is available.

1.3 DESCRIPTION OF DATASETS RELATED TO EXOTIC AFFORESTATION, REFORESTATION, AND DEFORESTATION

1.3.1 Balmoral (Waitaki) *P. nigra* (Afforestation Effects)

SUMMARY

This trial was established in 1993 as an agroforestry trial in a sub-humid zone in the Mackenzie Basin. It contains 4 replicates of 4 tree spacings (0, 250, 500, 750 sph) of *Pinus nigra* planted into undeveloped grassland. Half the trial has been sown with clovers and fertilised. Soil C was measured (0-0.3 m) between tree rows at year 5. As rows were 4 m apart and tree growth is slow in the environment this measurement was considered equivalent to time zero. Soil C was also measured at year 10 (in 2003). No change in soil C was evident. Above ground forest biomass was also measured at year 10. Results have been published.

DATABASE DESCRIPTION

Name of the dataset:	Balmoral (Waitaki) <i>P. nigra</i> Afforestation Effects
Primary contact:	Murray Davis, Scion
Data ownership / accessibility:	Scion, publicly available
Dataset type:	Experimental
Data storage:	Excel spreadsheet, Murray Davis
Broad Land use/Landform sampled:	Unimproved grassland and afforested unimproved grassland/High country terrace
Number of sites:	One
Geographical spread:	Trial site is located on Balmoral Station, South of Lake Tekapo, Mackenzie Basin
Soil orders:	Orthic Brown
Depth of sampling: 0-30 cm	Yes
Specific sampling depths:	0-10, 10-20, 20-30
Bulk Density measurements:	Yes
Multiple sampling through time:	Yes
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	Map coordinates
Soil description:	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	Yes
Land use history recorded:	Yes
If Yes above, then history for what period?	Approximately 100 yrs
Management history available:	
History timeframe:	Year-by-year data for 10 yrs proceeding last sampling, when the trial was planted, prior to that anecdotal history of grazing of unimproved grassland for previous 100 years
Management factors held:	Tree crop type, tree stocking rate, fertiliser type/rate and legume species introduction (for part of trial)
Data held in what form:	Excel spreadsheet

Mitigation opportunities to minimise or improve soil C:

Soil C has not changed in first 10 years since afforestation, is expected to increase as forest floor develops. Re-measurement might indicate that soil C could be increased through nitrogen input from legumes and fertiliser application.

Supplementary information on soils:

% total N for all plots, site data for pH, phosphorus (total, inorganic, organic, Bray-2, Olsen) and exchangeable cations.

Associated publications:

Davis M, Nordmeyer A, Henley D, Watt M. 2007. Ecosystem carbon accretion ten years after afforestation of depleted sub-humid grassland planted with three densities of *Pinus nigra*. *Global Change Biology* 13, 1414-1422.

Free-form database description**Origin:**

Above publication

Collection methodology:

Plot soil coring, initial sampling between tree rows (10 cores/plot), at year 10 random sampling along transects (25 cores/plot)

Sample analysis:

C and N analysis of < 2 mm mineral soil and pine and herbaceous roots in cores by LECO analyser

Strengths and limitations:

Strengths – replicated trial (4 reps), relatively uniform flat site, different tree stockings including unplanted control plots, unimproved and improved (fertiliser + legume) treatments, has tree productivity, biomass and carbon information, control over trial through joint venture ownership with land owner.

Limitations – Tree species is *Pinus nigra* rather than *Pinus radiata*

Assumptions and uncertainties:

Uncertainty of future funding provision

Main findings relevant to change in soil C stocks:

Afforestation at any tree stocking had no effect on soil C stocks at age 10, as expected for a low productivity semi-arid site

Future plans:

Trial re-measurement at 5-year intervals, next measurement 2008

1.3.2 Orton Bradley (Afforestation Effects)

SUMMARY

This trial was planted in pasture in 1999 at Orton Bradley Park on Banks Peninsula to compare the impact of 3 tree species (*Pinus radiata*, *Eucalyptus nitens*, *Cupressus macrocarpa*) on soil properties. The trial has 4 replicates. Soil was sampled (0-0.3 m) in 1999 and 2004 and is due for re-sampling at year 10.

DATABASE DESCRIPTION

Name of the dataset:	Orton Bradley Afforestation Effects
Primary contact:	Dr Leo Condron, Lincoln University
Data ownership / accessibility:	Dr Leo Condron, Lincoln University
Dataset type:	Experimental
Data storage:	Excel spreadsheets, L. Condron
Broad Land use/Landform sampled:	Pasture afforested in 1999/Hill country slopes
Number of sites:	One
Geographical spread:	Located at Orton Bradley Park, Banks Peninsula
Soil orders:	Pallic
Depth of sampling: 0-30 cm	Yes
Specific sampling depths:	0-5, 5-10, 10-20, 20-30 cm depths
Bulk Density measurements:	Yes
Multiple sampling through time:	Yes
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	Map coordinates
Soil description:	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	Yes
Land use history recorded:	Yes
If Yes above, then history for what period?	For 25+ yrs proceeding last sampling
Management history available:	
History timeframe:	Year-by-year data for 5 yrs since planting, anecdotal history of pasture grazing by sheep prior to planting
Management factors held:	Tree crop type, tree stocking rate, thinning and pruning history, fertiliser type/rate (nil since planting)
Data held in what form:	doc files

Mitigation opportunities to minimise or improve soil C:

Soil C is expected to decline following afforestation, and be mitigated by some amount with development of a forest floor

Supplementary information on soils:

% total N, total S, total, organic, inorganic and Olsen P, CEC and exchangeable cations for all samples

Associated publications: No

Free-form database description

Origin:

File notes

Collection methodology:

Soil coring

Sample analysis:

Full chemical analysis on < 2 mm soil

Strengths and limitations:

Strengths – Replicated trial (3 reps) designed to evaluate and compare the effect of three tree species on soil properties. Chronosequence sampling, sampled at time of planting and 5 year intervals.

Assumptions and uncertainties:

Uncertainty of future funding provision

Main findings relevant to change in soil C stocks:

There has been a reduction in soil C mass at year 5 of 2-5 Mg ha⁻¹. No statistical analysis has been undertaken at this stage.

Future plans:

Continued sampling at 5 year intervals

1.3.3 Afforested LTSP Series III Trials

SUMMARY

This trial series has been established at 14 sites that are expected to exhibit key nutrient deficiencies of radiata pine. Moisture availability is also limiting at some sites. The trials have been planted with different genotypes of *Pinus radiata* and *Cupressus* species, and of *Pseudotsuga menziesii* at two locations. The aim of the trials is to examine the genetic range of variability of response to limiting nutrient and moisture availability. At some of the sites in this series the trails were planted into pasture land.

DATABASE DESCRIPTION

Name of the dataset:	Afforested LTSP Series III Trials
Primary contact:	Dr Jianming Xue, Scion
Data ownership / accessibility:	Scion, publicly available
Dataset type:	Experimental
Data storage:	Jianming Xue Excel spreadsheets, shared drive Scion
Broad Land use/Landform sampled:	Recently afforested grassland/rolling and hill country
Number of sites:	3
Geographical spread:	Mahia Peninsula Gisborne, upper Rangitata catchment South Canterbury, Lawrence Central Otago
Soil orders:	Brown soils at 3 sites
Depth of sampling: 0-30 cm	No
Specific sampling depths:	Mahia: 0-5, 0-10, 0-20; Lawrence 0-10, 10-20; upper Rangitata 0-10, 10-20 cm depths
Bulk Density measurements:	No
Multiple sampling through time:	No
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	GPS
Soil description:	Yes for Mahia, no for other 2 sites
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	No
Land use history recorded:	Yes
If Yes above, then history for what period?	These are recently afforested x pasture sites - history is known accurately since planting (5 years)
Management history available:	
History timeframe:	Year-by-year data for 5 years since planting
Management factors held:	Tree species, tree spacing, fertiliser since planting (nil)
Data held in what form:	doc files
Supplementary information on soils:	
	Mahia – %C, total N, pH, for 0-5cm only; Lawrence – %C, %N and pH for all samples; upper Rangitata – no analyses so far.

Associated publications: None

Free-form database description

Origin:

As previously described

Collection methodology:

Soil coring, 25- 30 cores/plot

Sample analysis:

LECO

Strengths and limitations:

Replicated trials (4 reps) at each site. Soils sampled at time of planting. Effect of afforestation on soil properties with 2 species can be compared at upper Rangitata site, with 3 species at Lawrence site.

Assumptions and uncertainties:

Uncertainty of future funding

Main findings relevant to change in soil C stocks:

No main findings to date

Future plans:

Dependent on future funding.

APPENDIX 2: Datasets—Pastoral Agriculture And Soil Carbon

2.1 NATIONAL SOILS DATABASE

DATABASE DESCRIPTION

Name of the dataset:	NSD
Primary contact:	Allan Hewitt
Data ownership / accessibility:	Landcare Research
Data storage:	Electronic
Broad Land-use/Land form sampled:	all
Number of sites:	1500
Geographical spread:	NZ
Soil orders:	All
Depth of sampling: 0-30 cm -	yes
Specific sampling depths:	0-7.5, 0-10, 0-15, 7.5-15, 10-20, 15-30, 30-100 cm depths or sampling by horizon? Horizon
Bulk Density measurements:	less than half
Multiple sampling through time:	No
Pre-1990 soil sampling:	Yes
GPS reading of the site:	No
Soil description:	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	No
Land use history between sampling periods:	Yes
Intensity of land use:	Yes
Supplementary information on soils:	Yes
Management history	No

Mitigation opportunities to minimise or improve soil C:

No

Associated publications in journals, proceedings, Client reports to MfE, MAF, Regional Councils etc: Too many to list—see primary contact

2.2 NSD DEEP PROFILE RESAMPLING

SUMMARY

At March 2008, this dataset consisted of 66 re-sampled profiles collected around New Zealand. Landuses sampled include dairy, and a range of “drystock” land uses; drystock including sheep, beef, deer, horses dairy runoff etc. Most profiles were deeper than 60 cm, many closer to 1 m in depth. The data is held in an Excel spreadsheet by Louis Schipper at University of Waikato, there are plans to move this data into the National soils database.

This dataset has demonstrated that (i) dairy on flat land non-allophanic soils (19 profiles) have lost significant soil carbon (about $1.0 \text{ t ha}^{-1} \text{ yr}^{-1}$) since first sampled, (ii) dairy on flat allophanic soils (13 profiles), “drystock” on flat land non-allophanic soils (23 profiles) and “drystock” on allophanic soils (2 profiles) have not changed in soil C status; and, (iii) “drystock” on hill country (8 profiles) have gained soil C .

DATABASE DESCRIPTION

Name of the dataset:	NSD deep profile re-sampling
Primary contact:	Associate Professor Louis Schipper University of Waikato
Data ownership / accessibility:	Landcare Research, University of Waikato
Data storage:	Data is currently held in an Excel spreadsheet (Louis Schipper, University of Waikato). Copies are also held by co-workers. There are plans to integrate data into National Soils Database
Broad Land-use, Landform sampled:	Pasture (mainly drystock, dairy), Hill and flat land
Number of sites:	As at March 2008, 66 profiles sampled
Geographical spread:	North and South Island
Soil orders:	Allophanic (15), Brown (16), Gley (10), Granular (1), Melanic (3), Oxidic (1), Pallic (7), Pumice (1), Recent (6), Semiarid (5), Ultic (1).
Funding:	FRST programmes: Soil services and SLURI.
Depth of sampling: 0-30 cm:	Yes
Specific sampling depths:	Sampled by horizon to at least 60 cm
Bulk Density measurements:	Yes
Multiple sampling through time:	Yes
Pre 1990 soil sampling:	Yes
Site information available	Yes
Information type:	GPS coordinates
Soil description	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	Yes
Land use history recorded:	Yes
If yes above then history for what period:	land use/management recorded at both sampling times (about 20 years apart), some information about intervening history depending on farmer availability and knowledge

Management history available	Not detailed, see above
History timeframe:	Patchy. See above
Management factors held:	Patchy. See above
Data held in what form:	Written notes some in spreadsheets

Mitigation opportunities to minimise or improve soil C:

This data set will identify broad management practices that are either gaining or losing soil C to refine/ eliminate hypothesis and then develop appropriate strategies to enhance C uptake or reduce losses.

Supplementary information on soils: Total N, some Olsen P data available

Associated publications:

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

Free-form database description

Origin:

This data set was originally started to determine the rates of net N immobilization in New Zealand soils. The purpose of this dataset has now been expanded to also determine the size of change in soil C and N for a range of land use, land forms and soil orders in New Zealand.

Collection methodology:

A full description of the sampling approach is provided in Schipper et al. (2007). Briefly, sites were initially selected that had been originally sampled about 17+ years previously, were re-locatable based on field notes taken at the time of original sampling, had archived soil samples that could be re-analysed, and had bulk density information available. Sites excluded those with buried top-soils or having peaty top-soils or had not been in pasture when first sampled. At times, selected sites were not sampled or included in the dataset where the exact site could not be located once in the field (such as a poor match with original profile description), sites had been obviously modified by construction of roads or buildings. Brief site history was obtained from farmers where possible, this was often not particularly detailed as farms had changed hands or there were no written records.

Once the original sample site has been relocated a pit is dug and horizon depths compared to records. Sampling is exactly matched to the sampling previous sampling. In general, a single soil sample is taken from each horizon for chemical analysis and bulk density cores were taken from the centre of each horizon. In the past, for some sites, bulk density cores were not taken from the centre of the horizon as the purpose for sampling changed.

The method for collecting bulk density samples has changed from original sampling. Historically, soil cores were taken using a 200-core sampler (Soil Moisture Equipment Corp, Santa Barbara, California). This method was originally designed to obtain soil cores to determine moisture release curves which also allowed bulk density to be calculated. For contemporary sampling, the cores were carved into the centre of the horizon using a sharp knife, this technique that generally avoids shattering the core. A correction factor was developed to account for differences between these sampling methods and demonstrated that bulk density had not changed between sampling times (Schipper et al., 2007).

Sample analysis:

Soil samples were air-dried and analysed for total C and N using a LECO furnace. Archived soils samples were retrieved and also analysed in the same LECO run to minimize laboratory errors.

Main findings relevant to change in soil C stocks:

Further profiles continue to be collected but results from statistical analysis on the dataset as at March 2008 are presented in Figure 2.1. Main findings for the top meter of the profile were:

- (i) dairy on flat land non-allophanic soils (19 profiles) have lost significant soil carbon (about $1.0 \text{ t ha}^{-1} \text{ yr}^{-1}$) since first sampled,
- (ii) dairy on flat allophanic soils (13 profiles), “drystock” on flat land non-allophanic soils (23 profiles) and “drystock” on allophanic soils (2 profiles) have not changed in soil C status; and,
- (iii) “drystock” on hill country (8 profiles) have gained soil C .

Statistical analysis on just the top 30 cm.

Future plans:

Sampling to continue to better partition C change (loss or gain) by land use/land form/soil order. We plan to submit a paper to a refereed journal by June 2009.

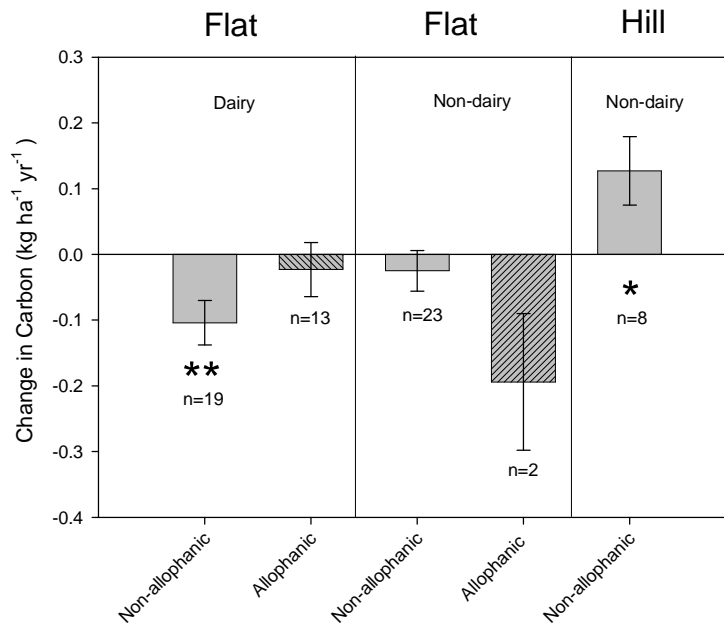


Figure 2.2. Annual rate of change in soil C stocks in the top metre during past 17+ years. Profiles are split into combinations of major landuse, land form and allophanic/non-allophanic soils. n is the number of profiles for each combination, * and ** means significantly different from zero at $P < 0.05$ and $P < 0.01$, respectively.

2.3 AGRESEARCH SOIL BIOINDICATOR DATASET

SUMMARY

In 1995 AgResearch started a FRST-funded project to better quantify soil biological and biochemical characteristics. This project was continued for five years. Normal protocols of soil testing for pasture sector were followed i.e. soils were collected from 0-7.5 cm depth and troughs, fence line, dung and recent urine patches were avoided during sampling. Soil samples were collected mainly from pastures that were on ash, sedimentary and pumice soils. Some soil samples were also collected from nearby cropping, forestry and market gardening sites to compare the soil biological and biochemical characteristics. In 2006 and 2008 some of sites in the Waikato and Northland were re-sampled. Results of soil carbon levels from the Waikato soils under pasture land use have now been analysed. On average, in comparison to 1995-96, the amounts of soil carbon in these re-sampled pasture sites appeared to have increased in the top 0-7.5 cm depth (Figure 2.2).

DATABASE DESCRIPTION

Name of the dataset:	Bioindicator soil data set
Primary contact:	Anwar Ghani
Data ownership / accessibility:	AgResearch
Data storage:	Data is currently held in an Excel spreadsheet and Access Database format
Broad Land-use, Landform sampled:	Pasture (mainly dairy and sheep/beef), Rolling hill and flat land
Number of sites:	136 sites
Geographical spread:	North (110) and South Island (26)
Soil orders:	Allophanic (64), Brown (36), Pumice (36),
Funding:	FRST programmes: FORST funded: Bioindicator (1995-1998) and SLURI (2006-current).
Depth of sampling: 0-30 cm:	No
Specific sampling depths:	0-7.5 cm
Bulk Density measurements:	Yes
Multiple sampling through time:	Yes (seasonal sampling for couple of years on 36 sites)
Pre 1990 soil sampling:	No
Site information available	Yes
Information type	GPS coordinates
Soil description	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	Yes (only 36 sites were re-sampled after 10 years gap)
Land use history recorded:	Yes
If yes above then history for what period?	land use/management recorded at the sampling times (about 10 years ago)
Management history available	Most sites sampled have been under pasture land use for over 30 years.
History timeframe:	See above

Management factors held:	Good record of fertilizer inputs and grazing intensity is available at most sites.
Data held in what form:	Excel spreadsheets and Access Data base

Mitigation opportunities to minimise or improve soil C:

Very little. Some of the re-sampled sites from this data set will identify if soil carbon in the top 0-7.5 cm depth has increased or decreased over the last 10 years. It will be possible to make some comments on effects of increased grazing intensity on the soil carbon in the ash soils.

Supplementary information on soils e.g. %total N, pH, production:

Total N, Olsen P, total S, microbial biomass-C and Hot-water extractable C data is also available on most of the sites that were sampled originally between 1995-1998 period.

Associated publications:

- Ghani A, Sarathchandra U, Perrot KW, Wardle DA, Singleton P, Dexter M (1996) Biological and biochemical quality of pastoral soils: special and temporal variability. *Proceedings of the New Zealand Grassland Association* 58: 211–218.
- Ghani A, Dexter M, Perrott KW (2003) Hot-water extractable carbon in soils: a sensitive measurement for determining impacts of fertilisation, grazing and cultivation. *Soil Biology and Biochemistry* 35: 1231–1243.
- Ghani A, Dexter M, Carran RA, and Theobald TW (2007) Dissolved organic nitrogen and carbon in pastoral soils: the New Zealand experience. *European Journal of Soil Science* 58: 832–843.

Free-form database description

Origin:

This data set was originally started to better quantify ranges of the soil biological and biochemical properties in the high producing dairy and sheep/beef farms in New Zealand. The purpose of this dataset was to identify soil biochemical indicators that would respond to changes in pasture management including fertilization, grazing intensity and cultivation. Hot-water carbon pool in soils was identified to be one of the better indicators of changes in soil management.

Collection methodology:

A full description of the sampling approach is provided in Ghani et al. (2007). Briefly, the monitoring sites were selected based on the milk solid production or stock carrying capacity. Potential sites were visited initially and based on the identification of soil type and uniformity of soil profile up to a depth of 60 cm, an average paddock of the farm was selected for sampling. Most of the dairy pastures were located mainly on flat to rolling land and sheep/beef farms were located on predominantly rolling to steep land. Sixty soil cores (2.5 cm diameter and 7.5 cm depth) on transect of 100 m were collected from each of the monitoring sites and bulked together. Bulk densities from 0-5 cm depth were also measured at most of the monitoring sites.

Sample analysis:

All the soil biochemical properties were determined using field moist samples. Total C, N and S were measured in the air-dried samples using a LECO furnace.

Main findings relevant to change in soil C stocks:

The re-sampling of the original sites has been limited to Ash soils in the Waikato area only. Another re-sampling has been completed in the Northland area but results have not been analysed yet. Based on the results of soil carbon in the top 0-7.5 cm depth, over all the allophanic soils appears to have gained both total C and total N in soils over the last 10 years.

Future plans:

Sampling of other pastoral soils will continue that were initially sampled 10-12 years ago to better understand changes in soil C stock.

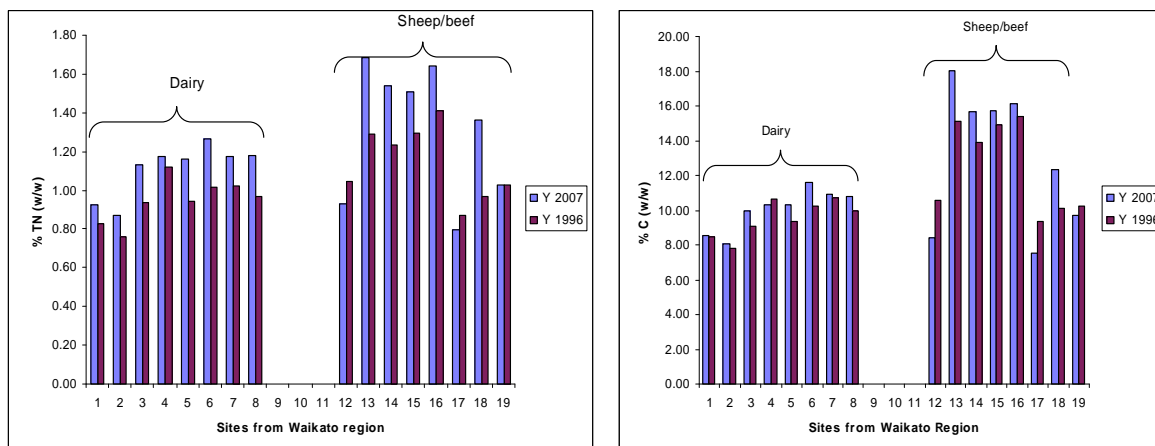


Figure 2.2 Results of re-sampling pasture sites in the Waikato region.

2.4 WHATAWHATA ARCHIVED SOIL SAMPLES

SUMMARY

To determine whether carbon was being lost in hill country pastures, we analysed archived soil samples collected between 1984 and 2006 from two slope classes (steep and easy) at the Whatawhata Hill Country Research Station. Soil samples had been collected from paddocks that were fertilized with six different loading rates of P (0 to 100 kg ha⁻¹ yr⁻¹), the primary limiting nutrient for grass-clover pastures in these hill country farms. Soils are archived by AgResearch (Hamilton) and have been analysed by Louis Schipper (University of Waikato). The range of P fertilizer loadings allowed us to determine whether P would regulate changes soil C and N. In contrast to expectations, there was no unidirectional change in C and N between 1984 and 2006 and size of changes in C and N were not dependent on P loading rate. On average, soil C initially increased during the first 6 years of the trial at 0.27 % C yr⁻¹ (1.56 t ha⁻¹ yr⁻¹) and 0.156 % C yr⁻¹ (1.06 t ha⁻¹ yr⁻¹) on easy and steep slopes, respectively. Subsequently, soil C declined at -0.024 % yr⁻¹ for the easy slopes (not significantly different from 0) and -0.066 % yr⁻¹ (0.45 t ha⁻¹ yr⁻¹) for the steep slopes. Similarly, % N increased between 1984 and 1990 at 0.025% N yr⁻¹ (144 kg ha⁻¹ yr⁻¹) and 0.012 % N yr⁻¹ (82 kg ha⁻¹ yr⁻¹) on easy and steep slopes, respectively. Post-1990, small but significant, losses of total N were measured on the steep slopes of 0.004% yr⁻¹ (27 kg N ha⁻¹ yr⁻¹) with no change on the easy slopes. Differences in pasture production are the most likely explanation for the changes in total C and N. Post 1990, there was a decrease in pasture dry matter production in summer/early autumn (about 40% less) and declines in N-fixing clover abundance. Rainfall was greater pre-1990 than post-1990 during these seasons and it appeared that post-1990 pasture production was limited by moisture rather than phosphorus. This study is currently being written up as a paper for submission to a refereed journal (planned submission date June 2008).

DATABASE DESCRIPTION

Name of the dataset:	Whatawhata – P trial
Primary contact:	Associate Professor Louis Schipper University of Waikato
Data ownership / accessibility:	AgResearch, University of Waikato
Data storage:	Data is currently held in an Excel spreadsheet (Louis Schipper, University of Waikato).
Broad Land-use, Landform sampled:	Pasture, sheep, Hill Country
Number of sites:	12 paddocks, with 6 P fertiliser rates, 2 replicate paddock per P loading
Geographical spread:	North Island
Soil order:	The soils were a soil association of Typic Impeded Allophanic Soils (Dunmore series) and Typic Orthic Granular Soils (Naikē series) on the easy slopes, and Typic Yellow Ultic Soils (Kaawa series) on the steeper slopes.
Funding:	FRST programmes: Soil services
Depth of sampling: 0-30 cm:	No
Specific sampling depths:	0-7.5 some 0-15 cm also available but not currently analysed
Bulk Density measurements:	Yes, but only for 2007

Multiple sampling through time:	10 sampling times in 1984-2007
Pre-1990 soil sampling:	Yes
Site information available	Yes
Information type	GPS coordinates, site map
Soil description	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	Yes
Land use history recorded:	Yes
If yes above then history for what period:	1984-2007 some information prior 1984
Management history available	Yes
History timeframe:	1984-2007
Management factors held:	Stocking rate, fertilizer rates
Data held in what form:	Written notes some in spreadsheets

Mitigation opportunities to minimise or improve soil C:

This data set will contribute to determining whether P addition to hill country pastures can increase or decrease soil C. This is one of few long-term records of changes in C and N in hill country.

Supplementary information on soils:

Total N, Olsen P data, quick test data also in spreadsheet held by AgResearch. Some pasture production data available. There may be more history information available in old records but these are not immediately available.

Associated publications:

Schipper, L.A.; Dodd, M.; Fisk, L.M.; Power, I.; Parenzee, J.; Arnold G. (in prep. For submission to *Biogeochemistry*) Soil carbon and nitrogen changes during 20 years of pasture development with different phosphorus fertiliser loadings.

Free-form database description

Origin:

This study was based at the Whatawhata Hill Country Research Station (2693705E, 6375215N), 22 km west of Hamilton, North Island, New Zealand on undulating to very steep hill country. The station is 810 ha in size, ranging from 45 to 370 m above sea level. The climate at the Research Station is mild to warm and humid, with a mean annual rainfall of approximately 1630 mm. A phosphate fertilizer trial was established at the site in 1980 on a 14.2 ha area subdivided into 20 paddocks of 0.25–1.22 ha in size, with easy to steep slopes (over 35°) and a north-westerly aspect. The soils were a soil association of Typic Impeded Allophanic Soils (Dunmore series) and Typic Orthic Granular Soils (Naike series) on the easy slopes, and Typic Yellow Ultic Soils (Kaawa series) on the steeper slopes.

Fertiliser application The trial site was converted to pasture from indigenous scrub and forest in the 1920s. The pasture was fertilized with single superphosphate (NPKS=0 9 0 11) at a rate of 36 kg P ha⁻¹ y⁻¹ for at least 12 years before the start of the trial. The fertiliser trial started in 1980 when five single superphosphate application rates were established: 10, 20, 30, 50 and 100 kg P ha⁻¹ y⁻¹ on four replicate paddocks. Fertiliser was applied in late summer/early autumn. After 1989, single superphosphate was replaced with triple superphosphate (NPKS=0 21 0 1). From 1985 to 2006, the P rate treatments continued only on two of the

replicate paddocks. Fertiliser application ceased on the other two replicate paddocks and we selected paddocks that had received $10 \text{ kg P ha}^{-1} \text{ y}^{-1}$ between 1980 and 1985 as nominal unfertilised controls ($0 \text{ kg P ha}^{-1} \text{ y}^{-1}$) for the current study.

Grazing management Between 1984 and 1988, paddocks were rotationally grazed by Romney-cross wethers or ewes from December to lambing (August) and set stocked through lambing to weaning (November). Pre-conditioning paddocks were used to minimize the transfer of nutrient via excreta from high P input to low P input paddocks. Stocking rates were adjusted to maintain pasture utilization across the P loading rates, from $\sim 12 \text{ SU ha}^{-1}$ on the unfertilized paddocks up to $\sim 18 \text{ SU ha}^{-1}$ on the highest fertilizer rate paddocks. From 1989-1991 the paddocks were continuously grazed with ewe hoggets and from 1991-1995 the grazing management reverted to that for the 1984-1988 period.

Pasture production Pasture growth rate data for easy slopes at Whatawhata were collated for the site based on several sources: a) between 1980-1988 pasture production was measured using a double-trim cage harvest technique on all paddocks in the fertilizer rate experiment, from which we selected data from the paddocks receiving $30 \text{ kg P ha}^{-1} \text{ y}^{-1}$; b) between 1988-1991 and 1993-1995 pasture production was measured using calibrated pre- and post-grazing visual assessments on adjacent north-facing easy paddocks also receiving $30 \text{ kg P ha}^{-1} \text{ y}^{-1}$; c) between 1997-1999 pasture production was measured using calibrated pre- and post-grazing radiometer measurements on paddocks within the fertiliser rate experiment; and d) between 2000-2004 pasture production was measured using the double-trim cage harvesting technique on adjacent north-facing easy paddocks also receiving $30 \text{ kg P ha}^{-1} \text{ y}^{-1}$. Net pasture growth rates for each season of each year were calculated by taking the mean daily growth rate for each month and averaging these values across the three months of each season. We nominated summer as January, February, and March to match summaries of climate data (see below).

Climate data Long term weather data was collected at the Research Station for the National Climate Database (NIWA), from which we selected monthly rainfall and mean air temperature data for the period 1984-2007. In this region, pasture growth is mostly sensitive to rainfall during the late summer-autumn period (January – March) where precipitation-evapotranspiration deficits are exacerbated by slope, while during the rest of the year soils are not usually in moisture deficit (Bircham & Gillingham 1986). Consequently, we calculated seasonal departures from long-term mean rainfall for summer to determine whether there had been a multi-year climate change pattern during the study.

Collection methodology:

Soil Sampling Throughout the trial, soil samples were collected from all paddocks in February or March of each year. The sampling methodology changed during the course of the trial. Between 1983 and 1988, four soil cores were taken along each of five transects on both easy ($10\text{-}20^\circ$) and steep ($30\text{-}40^\circ$) slopes classes, from each paddock (20 replicate cores per paddock and slope class). The cores were sectioned into 0-30 mm and 30-70 mm bulked by depth for each transect. For samples collected between 1993 and 2006, 15-20 soil cores (0-75 mm) were randomly taken from easy and steep slopes of each paddock, and then bulked by each paddock's slope class. All soils were air-dried, passed through a 2-mm sieve and stored in plastic containers. For the years when samples were collected by transect (1983-1988) we bulked sub-samples from the five transects to give a single sample for each depth from each paddock/year and slope combination before carbon and nitrogen analysis (see below).

Bulk density Intact soil samples for bulk density determination were collected at the end of the trial. Three replicate stainless steel rings (100 mm diameter, 75 mm high) were hand carved and pressed into the soil of each paddock/slope combination (Cook et al. 1993). These soil samples were dried to constant weight at 105°C and dry bulk density calculated.

Sample analysis:

Soil Analysis Prior to C and N analysis, any large visible fragments of roots and grass were removed from air-dried soil and the sample then ground in a ball mill before total C and N analysis using a LECO furnace (TruSpec, St Joseph, Mississippi). Data were corrected for moisture factors which were obtained for each air-dried soil sample following drying to a constant weight at 105°C.

Main findings relevant to change in soil C stocks:

In contrast to expectations, there was no unidirectional change in C and N between 1984 and 2006 and size of changes in C and N were not dependent on P loading rate. On average, soil C initially increased during the first 6 years of the trial at 0.27 % C yr⁻¹ (1.56 t ha⁻¹ yr⁻¹) and 0.156 % C yr⁻¹ (1.06 t ha⁻¹ yr⁻¹) on easy and steep slopes, respectively. Subsequently, soil C declined at -0.024 % yr⁻¹ for the easy slopes (not significantly different from 0) and -0.066 % yr⁻¹ (0.45 t ha⁻¹ yr⁻¹) for the steep slopes (Table 1). Similarly, % N increased between 1984 and 1990 at 0.025% N yr⁻¹ (144 kg ha⁻¹ yr⁻¹) and 0.012 % N yr⁻¹ (82 kg ha⁻¹ yr⁻¹) on easy and steep slopes, respectively. Post-1990, small but significant, losses of total N were measured on the steep slopes of 0.004% yr⁻¹ (27 kg N ha⁻¹ yr⁻¹) with no change on the easy slopes. Differences in pasture production are the most likely explanation for the changes in total C and N (Table 2). Post 1990, there was a decrease in pasture dry matter production in summer/early autumn (about 40% less) and declines in N-fixing clover abundance. Rainfall was greater pre-1990 than post-1990 during these seasons (Figure 2.3) and it appeared that post-1990 pasture production was limited by moisture rather than phosphorus.

Future plans:

Planned submission of a completed paper by end of June 2008. Re-sampling of sites to deeper layers may be useful to determine whether a change in C and N in response to P loading is deeper in the profile at least to 50 cm. This would also provide an opportunity to match soils more carefully. Funding for maintaining the long-term management of Whatawhata Research station is not clear.

Table 2.1. Modelled % total C or N, the rate of change in total C and N, and the C:N at Whatawhata Research farm for both steep and easy slopes. Standard errors of rates of change are given in brackets. Bolded numbers were significant at 5% level.

Estimates	Total C		Total N		C:N ratio	
	Easy	Steep	Easy	Steep	Easy	Steep
Value at 1983 (%)	7.6	6.2	0.65	0.52	11.7	12.0
Total change	0.270	0.156	0.025	0.012	0.003	-0.002
Annual (% yr ⁻¹)	(0.030)	(0.029)	(0.003)	(0.003)	(0.026)	(0.025)
Value at 1989 (%)	9.3	7.2	0.80	0.60	11.7	11.9
Total change	-0.024	-0.066	0.000	-0.004	-0.045	-0.019
Annual (% yr ⁻¹)	(0.012)	(0.012)	(0.001)	(0.001)	(0.007)	(0.007)
Value at 2006 (%)	8.7	6.0	0.81	0.52	10.8	11.5

Table 2.2. Average dry matter production (kg DM ha⁻¹ d⁻¹) on easy slopes from averaged across all paddocks. Standard errors in brackets.

Season	Average for 1980 to 1989	Average for 1990 to 2004	P value for difference
Summer (Jan-Mar)	38 (3)	21 (3)	0.002
Autumn (April-June)	22 (2)	17 (2)	0.057
Winter (July-Sept)	26 (2)	28 (2)	0.42
Spring (Oct-Dec)	60 (3.5)	52 (2)	0.49

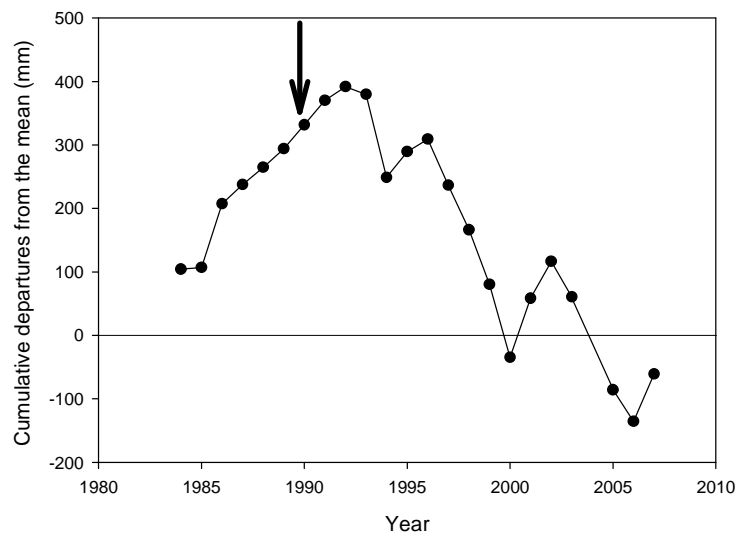


Figure 3.3. Cumulative departures from the mean of rainfall during summer (January – March) demonstrating that 1980s were wetter than 1990s at the site. Arrow indicates time when total C and N in top soil switched from accumulation to losses.

2.5 AGRESEARCH WINCHMORE LONG-TERM FERTILISER EXPERIMENT (WM1/1)

SUMMARY

The P and S fertiliser experiment (Nguyen et al., 1989) at Winchmore on grazed, border-strip irrigated pasture was initiated in 1952. The current treatments are: i) Control, no fertiliser applied; ii) 188 kg single superphosphate ha⁻¹ yr⁻¹; iii) 250 kg single superphosphate ha⁻¹ yr⁻¹; iv) 376 kg single superphosphate ha⁻¹ yr⁻¹; v) Sechura RPR plus elemental S equivalent to P and S in 250 kg single superphosphate ha⁻¹ yr⁻¹.

Lime was applied to the site in 1949, 1950 (both 2.5 t ha⁻¹) and in 1972 (4.4 t ha⁻¹). The site has not been cultivated since 1950.

There was an initial increase in soil C in the P and S fertiliser experiment at Winchmore from 1952 to 1963, irrespective of superphosphate application (Nguyen and Goh, 1990). This was associated with increased herbage production because of irrigation, pasture re-sowing and liming, and followed the effective loss of C in topsoil when the paddocks were border-dyked (using a grader). Over the initial period soil C, in the surface 7.5cm in the fertilised treatments, increased from 2.7% to 3.7%, a sequestration rate of 0.9 t C ha⁻¹ yr⁻¹. For the next 30 years soil C changed little with more surface soil C in treatments receiving superphosphate than in the unfertilised treatment (Nguyen and Goh, 1990). However by 1993 researchers (Murata *et al.*, 1995; Olsen, 1994) found that there was little difference in soil C between treatments. A trend for the 188 kg superphosphate ha⁻¹ yr⁻¹ treatment to have more soil C than the 0 and 376 kg superphosphate ha⁻¹ yr⁻¹ treatments is apparent in 0-7.5cm samples from 1993 to 2001 (Mean soil C for 4 sampling dates: 3.95, 4.13, 3.94% C for 0, 188 and 376 kg superphosphate ha⁻¹ yr⁻¹ respectively; LSD_{0.05} 0.17%). This result could be because of under-utilisation of herbage by stock on the 188 kg superphosphate ha⁻¹ yr⁻¹ treatment (the stocking rate on this treatment was reviewed in 1996 and increased). However Stewart and Metherell (1999a) found no significant differences between treatments in soil C at 0-10cm and 10-20cm depths in 1997. There were no irrigation or fertiliser effects on soil bulk density at Winchmore (Stewart and Metherell, 1999a), so differences in soil C between treatments would be proportionately similar whether expressed as a percentage or on an area basis.

Comparisons have also been made with a nearby dryland, ungrazed “wilderness site” (Haynes and Williams, 1992; Olsen, 1994). There have also been studies of organic matter physical fractions, microbial biomass and C14 ages on this site.

DATABASE DESCRIPTION

Name of the dataset:	Winchmore Long-term Fertiliser Experiment
Primary contact:	Anwar Ghani
Data ownership / accessibility:	AgResearch
Data storage:	Data is currently held in an Excel spreadsheets. Pasture production and soil test data up to 1992 in SAPT database.
Broad Land-use, Landform sampled:	Pasture (border-dyke irrigation)
Number of sites:	1 site
Geographical spread:	Winchmore Irrigation Research Station, Mid-Canterbury

Soil orders:	Brown
Funding:	FRST-funded until about 2002. AgResearch and FertResearch funding since.
Depth of sampling: 0-30 cm:	No
Specific sampling depths:	0-7.5 cm (regular soil testing, archived samples); 0-10cm and 10-20 cm occasional sampling.
Bulk Density measurements:	Yes (occasional)
Multiple sampling through time:	Yes (seasonal sampling 0-7.5 cm for most years – 4 * per year)
Pre 1990 soil sampling:	Yes
Site information available:	Yes
Information type:	Map & GPS coordinates easily obtained.
Soil description	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	Yes as % C in 0-7.5 cm converted to kg C / m ² using assumed bulk density.
Land use history recorded:	Yes
If yes above then history for what period?	Land use/management has been nearly constant throughout trial history with records kept.
Management history available:	Yes
History timeframe:	1948 - present
Management factors held:	Fertilizer inputs and grazing management.
Data held in what form:	Excel spreadsheets and SAPT Access Data base

Mitigation opportunities to minimise or improve soil C:

Very little. Over the long-term Superphosphate fertiliser has had no effect on soil C levels.

Supplementary information on soils:

Pasture production continuously, clover / grass / weeds %, Total N, total S, total P, Quick tests (pH, Olsen P, K, Mg, Ca) are available for most of the duration of the trial. SO₄-S since 1989. Extractable Organic S since 1995, microbial biomass-C and Hot-water extractable C, data in occasional measurements.

Associated publications:

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Free-form database description

Origin:

Experiment initiated in 1952 at Winchmore Irrigation Research Station in mid-Canterbury. Soil samples have been collected and archived on a regular basis since 1958. Archived samples from 1952 to 1986 were analysed for total C by Nguyen and Goh (1990). More recent samples from this set have also been analysed for total C (Metherell, 2003) as well as a

number of separate studies using samples specifically collected over greater depths at single points in time.

Collection methodology:

Mostly 0-7.5 cm samples collected with 2.5cm diameter corer. Approximately 15 cores from 0.1 ha plots.

Sample analysis:

Analyses presented by Nguyen and Goh (1990) used Walkley – Black, Olsen (1994) used mass-spec, while recent studies by Stewart and Metherell used LECO CNS.

Main findings relevant to change in soil C stocks:

Results from the Winchmore long-term rates of superphosphate fertiliser experiment showed that pasture development and irrigation resulted in an increase in soil C, but it must be noted that this was for initial development of a recently border-dyked site. There had been considerable soil disturbance prior to the beginning of the experiment and soil C levels were unnaturally low. The initial rate of increase was equivalent to about 1.5 t C ha⁻¹ yr⁻¹. A steady-state level was reached about 15 years after pasture establishment. These results indicated that topsoil C content was slightly higher in the fertilised plots than in unfertilized plots, but increasing the rate of Superphosphate from 188 to 376 kg ha⁻¹ gave no further increase.

Longer term equilibrium soil C on the Winchmore long-term fertiliser trial show no effect of fertiliser treatment on soil C levels (Stewart and Metherell, 1999a; Metherell, 2003), despite the large differences in herbage production. This, in part, results from the greater proportional allocation of C to plant roots in the absence of superphosphate so that there is much less difference in root production between treatments (Stewart and Metherell, 1999b; Metherell, 2003). Plant material with a high lignin content and a low N concentration in the unfertilised treatment (Metherell, 2003) will decompose more slowly. Differences in the quality of plant herbage and roots arise from both the direct effects of nutrient availability and the influence of nutrient availability on botanical composition. The decomposition rate would also be slower because of the lower earthworm biomass when superphosphate was not applied (Fraser *et al*, 1994).

There is also little difference between the soil C levels on the grazed and irrigated trial site and the ungrazed, unirrigated, unfertilized ‘wilderness area’.

Future plans:

Nil.

There should be reanalysis of the complete historical series by one analytical method to resolve some discrepancies between the results presented by Nguyen and Goh (1990) and more recent analyses of the 0-7.5 cm samples. More and secure funding is required to maintain this experiment.

2.6 AGRESEARCH WINCHMORE LONG-TERM IRRIGATION EXPERIMENT (WM4/1)

SUMMARY

The long-term irrigation frequency experiment (Rickard and McBride, 1986) at Winchmore was initiated in 1949 on border-strip irrigated pasture. Irrigation treatments were changed in 1953 and 1958, but dryland plots have never been irrigated and the site has not been cultivated since 1958. Treatments maintained since 1958 include: i) Dryland; ii) Irrigated at 10% soil moisture; iii) Irrigated at 20% soil moisture. Approximately 100 mm of water is applied per irrigation application. Superphosphate has been applied at 250 kg ha⁻¹ to all treatments annually. Lime was applied in 1948 (5.0 t ha⁻¹), 1953 (1.9 t ha⁻¹) and 1965 (4.1 t ha⁻¹).

On the Winchmore irrigation experiment analyses of soil C from around 1970 and from 1997 - 2001 have consistently shown a trend of highest C levels in the dryland treatment and significantly lower C levels in the most frequently irrigated treatment (20% soil moisture) (Metherell *et al.*, 2002; Stewart and Metherell, 1999a; Metherell, 2003), despite the increased herbage production with increasing irrigation frequency

DATABASE DESCRIPTION

Name of the dataset: Winchmore Long-term Irrigation Experiment

Primary contact:	Anwar Ghani
Data ownership / accessibility:	AgResearch
Data storage:	Data is currently held in an Excel spreadsheets. Pasture production up to 1995 in Access database.
Broad Land-use, Landform sampled:	Pasture (border-dyke irrigation)
Number of sites:	1 site
Geographical spread:	Winchmore Irrigation Research Station, Mid-Canterbury
Soil orders:	Brown
Funding:	FRST-funded until about 2002. AgResearch and FertResearch funding since.
Depth of sampling: 0-30 cm:	No
Specific sampling depths:	0-7.5 cm (annual soil testing); 0-10cm and 10-20 cm occasional sampling.
Bulk Density measurements:	Yes (occasional)
Multiple sampling through time:	Yes
Pre 1990 soil sampling:	Yes
Site information available	Yes
Information type:	Map & GPS coordinates easily obtained.
Soil description	Yes
Are soil samples archived?	Some
Total change in C or rate of change in soil C recorded:	No, as there are no initial soil samples available, and inconsistent depth of sampling.
Land use history recorded:	Yes
If yes above then history for what period?	

Land use/management has been nearly constant throughout trial history with records kept. Treatments constant since 1958.

Management history available

Yes

History timeframe:

1948 - present

Management factors held:

Irrigation management, Fertilizer inputs and grazing management.

Data held in what form:

Excel spreadsheets and SAPT Access Data base

Mitigation opportunities to minimise or improve soil C:

Very little. Over the long-term irrigation, especially in the most frequently irrigated regime, has resulted in lower soil C levels than dryland pasture under similar management.

Supplementary information on soils:

Pasture production continuously, clover / grass / weeds %, Soil moisture records, Quick tests (pH, Olsen P, K, Mg, Ca, SO₄-S) are available since 1981. Total N, total P, Extractable, microbial biomass-C and Hot-water extractable C data in occasional measurements.

Associated publications:

- Boul, H.L.; Garnham, M.L.; Hucker, D.; Baird, D.; Aislabie, J. 1994. Influence of agricultural practices on the levels of DDT and its residues in the soil. *Environmental Science and Technology* 28: 1397-1402.
- Condron, L.M.; Sinaj, S.; McDowell, R.W.; Dudler-Guela, J.; Scott, J.T.; Metherell, A.K. 2006. Influence of long-term irrigation on the distribution and availability of soil phosphorus under permanent pasture. *Australian Journal of Soil Research* 44: 127-133.
- Dudler, J. 2001. Influence of long-term irrigation on soil phosphorus availability and distribution under permanent grassland in New Zealand. Diploma, Swiss Federal Institute of Technology Zürich (ETHZ).
- Fitzgerald, P.D.; Rickard, D.S.; Mountier, N.S. 1963. Sampling errors associated with gravimetric soil moisture determinations. *New Zealand Journal of Agricultural Research* 6: 307-309.
- Fraser, P.M.; Piercy, J.E. 1996. Effects of summer irrigation on the seasonal activity, population size, composition and biomass of lumbricid earthworms in a long-term irrigation trial at Winchmore, New Zealand. *ASSSI and NZSSS National Soils Conference*. Volume 2: 89-90.
- Metherell, A.K.; Moss, R.A. 2002. Winchmore - 50 years of fertiliser and irrigation research. *New Zealand Society of Soil Science Golden Jubilee Conference 2002*.
- Metherell, A.K.; Stewart, D.P.C.; Carey, P.L.; Moss, R.A. 2002. Long-term irrigation improves soil quality, but decreases soil carbon sequestration. *Soil Quality and Sustainable Land Management Conference*. (Stephens, P., Callaghan, J., Austin, A. eds.) Landcare Research New Zealand Ltd. 55-61.
- Metherell, A.K. 2003. Management effects on soil carbon storage in New Zealand pastures. *Proceedings of the New Zealand Grassland Association* 65: 259-264.
- Rickard, D.S. 1968. Climate, pasture production and irrigation. *Proceeding of the New Zealand Grassland Association* 30: 81-93.
- Rickard, D.S.; Fitzgerald, P.D. 1970. The effect of soil moisture and irrigation on pasture production in Canterbury, New Zealand. *Proceedings of the XI International Grassland Congress*. 13-23.

- Rickard, D.S. 1972. Investigations into the response of pasture to irrigation, 1950-1957. Winchmore Irrigation Research Station. Technical Report No. 5.
- Rickard, D.S.; McBride, S.D. 1986. Irrigated and non-irrigated pasture production at Winchmore 1960 to 1985. *Winchmore Irrigation Research Station Technical Report 21*.
- Rickard, D.S.; McBride, S.D. 1986. Soil temperatures under non-irrigated and irrigated pasture. Winchmore Irrigation Research Station. Technical Report No. 20.
- Rickard, D.S.; McBride, S.D.; Fitzgerald, P.D. 1986. The effect of soil moisture deficits on pasture yield. *New Zealand Agricultural Science 20*: 7-12.
- Rickard, D.S.; McBride, S.D. 1987. Soil moisture records under irrigated and non-irrigated pasture. *Winchmore Irrigation Research Station Technical Report 23*.
- Stewart, D.P.C.; Metherell, A.K. 1996. Feeding the soil micro-organisms! - Using ^{13}C pulse labelling to investigate carbon cycling in pastoral systems. *ASSSI and NZSSS National Soils Conference*. 267-268.
- Stewart, D.C.P.; Metherell, A.K. 1998. Using ^{13}C pulse labelling to investigate carbon cycling in pastoral ecosystems. *16th World Congress of Soil Science*. Cirad. CD-ROM.
- Stewart, D.P.C.; Metherell, A.K. 1999. Long-term influence of management practices on the soil carbon of pastoral ecosystems. *Best Soil Management Practices for Production*. (Currie, L.D., Hedley, M.J., Horne, D.J., Loganathan, P. eds.) Fertilizer and Lime Research Centre, Massey University. Occasional Report No. 12: 317-326.
- Stewart, D.P.C.; Metherell, A.K. 1999. Carbon (^{13}C) uptake and allocation in pasture plants following field pulse labelling. *Plant and Soil 210*: 61-73.
- Stoker, R. 1982. Soil wetting and moisture extraction on a Lismore stony silt loam. Winchmore Irrigation Research Station. Technical Report No. 16. 14 pp.

Free-form database description

Origin:

Experiment initiated in 1948 at Winchmore Irrigation Research Station in mid-Canterbury, with the current treatments maintained since 1958. There have been sporadic measurements of soil carbon over the duration of the experiment.

Collection methodology:

Samples collected to various depths with 2.5cm diameter corer. Approximately 15 cores from 0.1 ha plots.

Sample analysis:

Analyses from 1956 to 1971 used Walkley – Black, while recent studies used LECO CNS.

Main findings relevant to change in soil C stocks:

On the Winchmore irrigation experiment analyses of soil C from around 1970 and from 1997 - 2001 have consistently shown a trend of highest C levels in the dryland treatment and significantly lower C levels in the most frequently irrigated treatment (20% soil moisture) (Metherell *et al.*, 2002; Stewart and Metherell, 1999a; Metherell, 2003), despite the increased herbage production with increasing irrigation frequency.

A number of factors combine to contribute to this observation. There is little difference in root production between treatments (Stewart and Metherell, 1999b; Metherell, 2003), lignin content of the litter is higher in the dryland treatment (Metherell, 2003) resulting in slower decomposition rates, lower pasture quality may have resulted in relatively more dung being

returned by the grazing animals, and there is more earthworm activity (Fraser, 1996) and higher microbial activity in moist soils.

Despite the decrease in soil carbon with increasing irrigation frequency there has been an improvement in soil quality (Metherell *et al*, 2002), especially an increased ability of the soil to store moisture, and a concomitant decrease in irrigation requirement.

Future plans:

Nil.

There should be a detailed investigation as to whether there has been a treatment effect on soil volume, which would affect the interpretation of the results of this experiment. Earthworm casting has resulted in a stone free surface layer in all treatments, but the depth to stones is about 10 cm under irrigation compared to 5 cm in the dryland treatment. To date it has been assumed that this is simply a redistribution of stones in the profile with no net change in soil volume.

More and secure funding is required to maintain this experiment.

2.7 AGRESEARCH TARA HILLS LONG-TERM GRAZING EXPERIMENT

SUMMARY

A long-term grazing experiment on a steep, oversown tussock site at Tara Hills High Country Research Station in the semi-arid (approx. 500 mm precipitation yr⁻¹) high country (910 m above sea level) of the South Island, New Zealand (Allan *et al.*, 1992) began in 1978. Most production data was collected in the first 10 years, with the grazing treatments being maintained since then, but AgResearch sold the Research Station to Ngai Tahu in 2007. The site is steep (27°) and contains indigenous short tussock species as well as improved legumes and grasses from oversowing. It has been fertilised with both P and S periodically since 1965. Treatments are continuous, alternating (two paddock system) or rotational (six paddock system) grazing with 1.9, 3.0 or 4.1 sheep ha⁻¹ during summer months. The experiment has a plot size of 1.7 ha and is unreplicated. The initial soil sampling in 1979 was of upper, mid and lower slope areas in the continuous treatment. In 1984 and 2003 intensive soil samplings of seven altitudinal strata within each plot were conducted. From 1996 to 1999 detailed studies of carbon cycling were conducted on the continuous and alternating grazing management treatments at three stocking rates. For statistical purposes the stocking rate by grazing management interaction is used as the error term, which gives a conservative assessment of statistical significance. For the 1984 and 2003 results a stocking rate by grazing management interaction term was estimated from the interaction of two orthogonal contrasts, with the remaining interaction terms used as the error term (Allan, 1985).

With pasture development, in a tussock grassland environment, soil C levels have in most treatments at least been maintained or possibly increased. In the 2003 soil sampling higher soil C levels were found in the stock camp zones at the upper part of each paddock of all treatments, but an altitudinal trend had not been observed for soil C in 1984 (Allan, 1985). In 2003, the lower stocking rate resulted in significantly higher soil C concentrations (Table 2), primarily because of high soil C levels in laxly grazed areas in the lowest altitudinal strata of some low stocking rate treatments. Over all altitudes the stocking rate effect was most pronounced in the continuous grazing management treatment, with the highest soil C levels found in the low stocking rate continuous grazing treatment, and the lowest soil C levels in the overgrazed high stocking rate continuous treatment. Although an effect of stocking rate was apparent in an initial pre-treatment sampling of the continuous treatment plots in 1979, and in the 1984 results (Allan, 1985) the magnitude of the effect has increased with time. Similar trends were observed in two samplings in 1997, particularly in the surface 10 cm, although the effect did not reach statistical significance (Stewart and Metherell, 2001).

DATABASE DESCRIPTION

Name of the dataset:	Tara Hills Long-term Grazing Experiment
Primary contact:	Anwar Ghani
Data ownership / accessibility:	AgResearch. Site now owned by Ngai Tahu
Data storage:	Excel spreadsheets
Broad Land-use, Landform sampled:	Pasture, Steep semi-arid hill country
Number of sites:	1 site
Geographical spread:	Tara Hills High Country Research Station, Omarama
Soil orders:	Brown
Funding:	FRST-funded until about 2002. Nil since.

Depth of sampling: 0-30 cm:	No
Specific sampling depths:	0-7.5 cm (annual soil testing); 0-10cm and 10-20 cm occasional sampling.
Bulk Density measurements:	Yes (1997)
Multiple sampling through time:	Yes
Pre 1990 soil sampling:	Yes
Site information available	Yes
Information type:	Map & GPS coordinates easily obtained.
Soil description	Yes
Are soil samples archived?	Some
Total change in C or rate of change in soil C recorded:	No, as there are no initial soil samples available, and inconsistent depth of sampling
Land use history recorded:	Yes
If yes above then history for what period?	Land use/management has been nearly constant throughout trial history with records kept Treatments constant since 1978.
Management history available	Yes
History timeframe:	1978 – present
Management factors held:	Grazing management.
Data held in what form:	Excel spreadsheets

Mitigation opportunities to minimise or improve soil C: Very little. Over the long-term higher stocking rates have resulted in lower soil C levels than laxly grazed treatments.

Supplementary information on soils:

Pasture production (early years of experiment), Occasional Quick tests (pH, Olsen P, K, Mg, Ca, SO₄-S). Total N, microbial biomass-C and Hot-water extractable C data in occasional measurements.

Associated publications:

- Allan, B. E. (1985). Grazing management of oversown tussock grassland in the South Island high country. PhD thesis, Lincoln College, Canterbury.
- Allan, B. E., O'Connor, K. F., and White, J. G. H. (1992). Grazing management of oversown tussock country 2. Effects on botanical composition. *New Zealand Journal of Agricultural Research* **35**, 7-19.
- Metherell, A.K. 2003. Management effects on soil carbon storage in New Zealand pastures. *Proceedings of the New Zealand Grassland Association* **65**: 259-264.
- Stewart, D.P.C.; Metherell, A.K. 1999b. Carbon (13C) uptake and allocation in pasture plants following field pulse labelling. *Plant and Soil* **210**: 61-73.
- Stewart, D.P.C.; Metherell, A.K. 2001. Influence of 20 years of different sheep grazing intensities and managements on soil organic matter quality in high country pastoral ecosystems; South Island, New Zealand. pp. 257-262 In: Understanding and managing organic matter in soils, sediments and waters. Eds. Swift, R.S., Spark, K.M. International Humic Substances Society. St. Paul.

Free-form database description**Origin:**

Experiment initiated in 1978 at Tara Hills Research Station near Omarama. Occasional measurements of soil C.

Collection methodology:

Samples collected to various depths with 2.5cm diameter corer. Approximately 15 cores per sample.

Sample analysis:

Originally Walkley – Black, while recent studies used LECO CNS.

Main findings relevant to change in soil C stocks:

Increased stocking rate is associated with a decrease in soil C compared to laxly grazed treatments, but with pasture development, soil C levels have in most treatments have at least been maintained or possibly increased

Future plans:

Nil.

2.8 AGRESEARCH WINCHMORE LONG-TERM FERTILISER EXPERIMENT

SUMMARY

The P and S fertiliser experiment (Nguyen et al., 1989) at Winchmore on grazed, border-strip irrigated pasture was initiated in 1952. The current treatments are:

- i) Control, no fertiliser applied;
- ii) 188 kg single superphosphate $\text{ha}^{-1} \text{yr}^{-1}$;
- iii) 250 kg single superphosphate $\text{ha}^{-1} \text{yr}^{-1}$;
- iv) 376 kg single superphosphate $\text{ha}^{-1} \text{yr}^{-1}$;
- v) Sechura RPR plus elemental S equivalent to P and S in 250 kg single superphosphate $\text{ha}^{-1} \text{yr}^{-1}$.

Lime was applied to the site in 1949, 1950 (both 2.5 t ha^{-1}) and in 1972 (4.4 t ha^{-1}). The site has not been cultivated since 1950.

There was an initial increase in soil C in the P and S fertiliser experiment at Winchmore from 1952 to 1963, irrespective of superphosphate application (Nguyen and Goh, 1990). This was associated with increased herbage production because of irrigation, pasture re-sowing and liming, and followed the effective loss of C in topsoil when the paddocks were border-dyked (using a grader). Over the initial period soil C, in the surface 7.5cm in the fertilised treatments, increased from 2.7% to 3.7%, a sequestration rate of $0.9 \text{ t C ha}^{-1} \text{yr}^{-1}$. For the next 30 years soil C changed little with more surface soil C in treatments receiving superphosphate than in the unfertilised treatment (Nguyen and Goh, 1990). However by 1993 researchers (Murata *et al.*, 1995; Olsen, 1994) found that there was little difference in soil C between treatments. A trend for the 188 kg superphosphate $\text{ha}^{-1} \text{yr}^{-1}$ treatment to have more soil C than the 0 and 376 kg superphosphate $\text{ha}^{-1} \text{yr}^{-1}$ treatments is apparent in 0-7.5cm samples from 1993 to 2001 (Mean soil C for 4 sampling dates: 3.95, 4.13, 3.94% C for 0, 188 and 376 kg superphosphate $\text{ha}^{-1} \text{yr}^{-1}$ respectively; $\text{LSD}_{0.05} 0.17\%$). This result could be because of under-utilisation of herbage by stock on the 188 kg superphosphate $\text{ha}^{-1} \text{yr}^{-1}$ treatment (the stocking rate on this treatment was reviewed in 1996 and increased). However Stewart and Metherell (1999a) found no significant differences between treatments in soil C at 0-10cm and 10-20cm depths in 1997. There were no irrigation or fertiliser effects on soil bulk density at Winchmore (Stewart and Metherell, 1999a), so differences in soil C between treatments would be proportionately similar whether expressed as a percentage or on an area basis.

Comparisons have also been made with a nearby dryland, ungrazed “wilderness site” (Haynes and Williams, 1992; Olsen, 1994). There have also been studies of organic matter physical fractions, microbial biomass and ^{14}C ages on this site.

DATABASE DESCRIPTION

Name of the dataset:	Winchmore Long-term Fertiliser Experiment
Primary contact:	Anwar Ghani
Data ownership / accessibility:	AgResearch
Data storage:	Data is currently held in Excel spreadsheets. Pasture production and soil test data up to 1992 in SAPT database.
Broad Land-use, Landform sampled:	Pasture (border-dyke irrigation)
Number of sites:	1 site

Geographical spread:	Winchmore Irrigation Research Station, Mid-Canterbury
Soil orders:	Brown
Funding:	FRST-funded until about 2002. AgResearch and FertResearch funding since.
Depth of sampling: 0-30 cm:	No
Specific sampling depths:	0-7.5 cm (regular soil testing, archived samples); 0-10cm and 10-20 cm occasional sampling.
Bulk Density measurements:	Yes (occasional)
Multiple sampling through time:	Yes (seasonal sampling 0-7.5 cm for most years – 4 * per year)
Pre 1990 soil sampling:	Yes
Site information available	Yes
Information type:	Map & GPS coordinates easily obtained.
Soil description	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	Yes, as % C in 0-7.5 cm converted to kg C / m ² using assumed bulk density
Land use history recorded:	Yes
If yes above then history for what period?	Land use/management has been nearly constant throughout trial history with records kept
Management history available	Yes
History timeframe:	1948 - present
Management factors held:	Fertilizer inputs and grazing management
Data held in what form:	Excel spreadsheets and SAPT Access Data base

Mitigation opportunities to minimise or improve soil C: Very little
Over the long-term Superphosphate fertiliser has had no effect on soil C levels.

Supplementary information on soils:

Pasture production continuously, clover / grass / weeds %, Total N, total S, total P, Quick tests (pH, Olsen P, K, Mg, Ca) are available for most of the duration of the trial. SO₄-S since 1989. Extractable Organic S since 1995, microbial biomass-C and Hot-water extractable C, data in occasional measurements.

Associated publications:

- AgResearch 1999. Accumulation of Uranium Through The Long Term Application of Superphosphate to Grazed Pastures at Winchmore. AgResearch.
- Boul, H.L.; Garnham, M.L.; Hucker, D.; Baird, D.; Aislabie, J. 1994. Influence of agricultural practices on the levels of DDT and its residues in the soil. *Environmental Science and Technology* 28: 1397-1402.
- Carey, P.L.; Metherell, A.K. 2003. Monitoring long-term changes in reserve potassium in some New Zealand soils using a modified sodium tetraphenyl-boron method. *New Zealand Journal of Agricultural Research* 46: 199–213.
- Condron, L.M. 1986. Chemical nature and plant availability of phosphorus present in soils under long-term fertilised irrigated pastures in Canterbury, New Zealand. PhD., University of Canterbury, New Zealand.

- Condrón, L.M.; Goh, K.M. 1989. Molecular weight distribution of soil organic phosphorus under irrigated pasture in New Zealand. *Journal of Soil Science* 40: 873-878.
- Condrón, L.M.; Goh, K.M. 1989. Effects of long-term phosphatic fertilizer applications on amounts and forms of phosphorus in soils under irrigated pasture in New Zealand. *Journal of Soil Science* 40: 383-395.
- Condrón, L.M.; Goh, K.M. 1990. Implications of long-term superphosphate applications on the accumulation and plant availability of soil phosphorus under irrigated pastures. *Proceedings of the New Zealand Grassland Association* 52: 191-193.
- Condrón, L.M.; Goh, K.M. 1990. Nature and availability of residual phosphorus in long-term fertilized pasture soils in New Zealand. *Journal of Agricultural Science, Cambridge* 114: 1-9.
- Condrón, L.M.; Sinaj, S.; McDowell, R.W.; Dudler-Guela, J.; Scott, J.T.; Metherell, A.K. 2006. Influence of long-term irrigation on the distribution and availability of soil phosphorus under permanent pasture. *Australian Journal of Soil Research* 44: 127-133.
- Curtin, D.; Beare, M.H.; McCallum, F.M. 2007. Sulphur in soil and light fraction organic matter as influenced by long-term application of superphosphate. *Soil Biology and Biochemistry* 39: 2547-2554.
- Edmeades, D.C.; Metherell, A.K.; Waller, J.E.; Roberts, A.H.C.; Morton, J.D. 2006. Defining the relationships between pasture production and soil P and the development of a dynamic P model for New Zealand pastures: a review of recent developments. *New Zealand Journal of Agricultural Research* 49: 207-222.
- Fraser, P.M.; Haynes, R.J.; Williams, P.H. 1994. Effects of pasture improvement and intensive cultivation on size of microbial biomass, enzyme activities and composition and size of earthworm populations. *Biology and Fertility of Soils* 17: 185-190.
- Goh, K.M.; Condrón, L.M. 1989. Plant availability of phosphorus accumulated from long-term applications of superphosphate and effluent to irrigated pastures. *New Zealand Journal of Agricultural Research* 32: 45-51.
- Gray, C.W.; McLaren, R.G.; Roberts, A.H.C.; Condrón, L.M. 1997. Effects of long-term phosphatic fertiliser applications on amounts and forms of cadmium in soils under irrigated pasture in New Zealand. *4th International Conference on the Biogeochemistry of Trace Elements*.
- Gray, C.W.; McLaren, R.G.; Roberts, A.H.C.; Condrón, L.M. 1999. The effect of long-term phosphatic fertiliser applications on the amounts and forms of cadmium in soils under pasture in New Zealand. *Nutrient cycling in agricultural ecosystems* 54: 267-277.
- Haynes, R.J.; Williams, P.H. 1992. Accumulation of soil organic matter and the forms, mineralization potential and plant-availability of accumulated organic sulphur: Effects of pasture improvement and intensive cultivation. *Soil Biology and Biochemistry* 24: 209-217.
- Haynes, R.J.; Williams, P.H. 1992. Long-term effect of superphosphate on accumulation of soil phosphorus and exchangeable cations on a grazed, irrigated pasture site. *Plant and Soil* 142: 123-133.
- McBride, S.D.; Nguyen, M.L.; Rickard, D.S. 1990. Implications of ceasing annual superphosphate topdressing applications on pasture production. *Proceedings of the New Zealand Grassland Association* 52: 177-180.
- McBride, S.D. 1992. Agronomic performance of RPR fertiliser in a grazed pasture. *Proceedings of the New Zealand Grassland Association*. 54: 41-44.
- McCall, D.G.; Thorrold, B.S. 1991. Fertiliser history is a useful predictor of soil fertility status. *Proceedings of the New Zealand Grassland Association* 53: 191-196.

- McDowell, R.W.; Condon, L.M. 2000. Chemical nature and potential mobility of phosphorus in fertilized grassland soils. *Nutrient Cycling in Agroecosystems* 57: 225-233.
- Metherell, A.K. 1992. Simulation of Soil Organic Matter Dynamics and Nutrient Cycling in Agroecosystems. PhD dissertation, Colorado State University.
- Metherell, A.K.; Cole, C.V.; Parton, W.J. 1993. Dynamics and interactions of carbon, nitrogen, phosphorus and sulphur cycling in grazed pasture. *Proceedings of the XVII International Grassland Congress*: 1420-1421.
- Metherell, A.K. 1994. A model for phosphate fertiliser requirements for pastures-incorporating dynamics and economics. *The efficient use of fertilisers in a changing environment: Reconciling productivity and sustainability*. (Currie, L.D., Loganathan, P. eds.) Fertiliser and Lime Research Centre, Massey University. Occasional Report No. 7: 18-37.
- Metherell, A.K. 1996. Costs and returns: Fertilisers. *Dairyfarming Annual* 48: 57-72.
- Metherell, A.K.; Condon, L.M.; Eden, R.M. 1997. Effect of fertiliser rates and forms on soil phosphorus fractions in a long term trial at Winchmore. *XVIII International Grassland Congress*. 1:10: 33-34.
- Metherell, A.; Condon, L.; Dudler, J.G.; Sinaj, S. 2001. Soil P build-up and the effect on P availability. News You Can Use. Vol. Summer 2001, pp. 10-11.
- Metherell, A.K. 2002. Management effects on soil carbon storage in New Zealand pastures. *Evaluation of Terrestrial Carbon Storage and Dynamics by In-situ and Remote Sensing Measurements*. 83-90.
- Metherell, A.K.; Moss, R.A. 2002. Winchmore - 50 years of fertiliser and irrigation research. *New Zealand Society of Soil Science Golden Jubilee Conference 2002*.
- Metherell, A.K. 2003. Management effects on soil carbon storage in New Zealand pastures. *Proceedings of the New Zealand Grassland Association* 65: 259-264.
- Murata, T.; Nguyen, M.L.; Goh, K.M. 1995. The effects of long-term superphosphate application on soil organic matter content and composition from an intensively managed New Zealand pasture. *European Journal of Soil Science* 46: 257-264.
- Nguyen, M.L.; Rickard, D.S.; McBride, S.D. 1989. Pasture production and changes in phosphorus and sulphur status in irrigated pastures receiving long-term applications of superphosphate fertiliser. *New Zealand Journal of Agricultural Research* 32: 245-262.
- 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.
- Nguyen, M.L.; Goh, K.M. 1992. Nutrient cycling and losses based on a mass-balance model in grazed pastures receiving long-term superphosphate applications in New Zealand 1. Sulphur. *Journal of Agricultural Science* 119: 107-122.
- Nguyen, M.L.; Goh, K.M. 1992. Nutrient cycling and losses based on a mass-balance model in grazed pastures receiving long-term superphosphate applications in New Zealand 1. Phosphorus. *Journal of Agricultural Science* 119: 89-106.
- Nguyen, M.L.; Goh, K.M. 1992. Status and distribution of soil sulphur fractions, total nitrogen and organic carbon in camp and non-camp soils of grazed pastures supplied with long-term superphosphate. *Biology and Fertility of Soils* 14: 181-190.
- Nguyen, M.L.; Goh, K.M. 1992. Sulphur mineralization and release of soluble organic sulphur from camp and non-camp soils of grazed pastures receiving long-term superphosphate applications. *Biology and Fertility of Soils* 14: 272-279.
- Olsen, C.D. 1994. The effect of soil fertility on organic matter fractions in a pastoral soil. B. Hort. Sc (Hons.) dissertation, Lincoln University.

- Olsen, C.D.; Metherell, A.K.; Goh, K.M. 1994. The effect of soil fertility on organic matter fractions in a pastoral soil. *New Zealand Soil News* 45: 152.
- Pamidi, J. 1999. Soil sulphur mineralization in relation to plant sulphur availability in soils under pastures. PhD., Lincoln University, New Zealand.
- Pamidi, J.; Goh, K.M.; McLaren, R.G. 2001. Comparison of three different methods of determining soil sulphur mineralization in relation to plant sulphur availability in soils. *Biology and Fertility of Soils* 34: 131-139.
- Quin, B.F. 1982. The quality of New Zealand superphosphate. *New Zealand Agricultural Science* 16: 93-100.
- Quin, B.F.; Rickard, D.S. 1983. Pasture production and changes in soil fertility on a long-term irrigated superphosphate trial at Winchmore, New Zealand. *Proceedings of the XIVth International Grassland Congress, Lexington, Kentucky, USA, June 15-24, 1981*: 323-326.
- Rickard, D.S. 1968. The requirements and residual effects of superphosphate on irrigated pasture. *Winchmore Irrigation Research Station Technical Report 2*.
- Rickard, D.S. 1968. Superphosphate requirements of irrigated pasture. *Winchmore Irrigation Research Station Technical Report 1*.
- Rickard, D.S.; McBride, S.D. 1987. Long term application and residual effects of superphosphate and effects of reactive phosphate rock on irrigated pasture. 53 pp.
- Stewart, D.P.C.; Metherell, A.K. 1996. Feeding the soil micro-organisms! - Using ^{13}C pulse labelling to investigate carbon cycling in pastoral systems. *ASSSI and NZSSS National Soils Conference*. 267-268.
- Stewart, D.C.P.; Metherell, A.K. 1998. Using ^{13}C pulse labelling to investigate carbon cycling in pastoral ecosystems. *16th World Congress of Soil Science*. Cirad. CD-ROM.
- Stewart, D.P.C.; Metherell, A.K. 1999. Long-term influence of management practices on the soil carbon of pastoral ecosystems. *Best Soil Management Practices for Production*. (Currie, L.D., Hedley, M.J., Horne, D.J., Loganathan, P. eds.) Fertilizer and Lime Research Centre, Massey University. Occasional Report No. 12: 317-326.
- Stewart, D.P.C.; Metherell, A.K. 1999. Carbon (^{13}C) uptake and allocation in pasture plants following field pulse labelling. *Plant and Soil* 210: 61-73.
- Villalobos, F.J.; Goh, K.M.; Saville, D.J.; Chapman, R.B. 1997. Interactions among soil organic matter, levels of the indigenous entomopathogenic bacterium *Serratia entomophila* in soil, amber disease and the feeding activity of the scarab larva of *Costelytra zealandica*: a microcosm approach. *Applied Soil Ecology* 5: 231-246.
- Williams, P.H.; Haynes, R.J. 1990. Cycling of P and S through the soil-plant-animal system under intensively grazed grass-clover pastures. *Proceedings of the 14th International Soil Science Congress*: 276-281.
- Williams, P.H.; Haynes, R.J. 1992. Balance sheet of phosphorus, sulphur and potassium in a long-term grazed pasture supplied with superphosphate. *Fertilizer Research* 31: 51-60.

Free-form database description

Origin:

Experiment initiated in 1952 at Winchmore Irrigation Research Station in mid-Canterbury. Soil samples have been collected and archived on a regular basis since 1958. Archived samples from 1952 to 1986 were analysed for total C by Nguyen and Goh (1990). More recent samples from this set have also been analysed for total C (Metherell, 2003) as well as a

number of separate studies using samples specifically collected over greater depths at single points in time.

Collection methodology:

Mostly 0-7.5 cm samples collected with 2.5cm diameter corer. Approximately 15 cores from 0.1 ha plots.

Sample analysis:

Analyses presented by Nguyen and Goh (1990) used Walkley – Black, Olsen (1994) used mass-spec, while recent studies by Stewart and Metherell used LECO CNS.

Main findings relevant to change in soil C stocks:

Results from the Winchmore long-term rates of superphosphate fertiliser experiment showed that pasture development and irrigation resulted in an increase in soil C, but it must be noted that this was for initial development of a recently border-dyked site. There had been considerable soil disturbance prior to the beginning of the experiment and soil C levels were unnaturally low. The initial rate of increase was equivalent to about $1.5 \text{ t C ha}^{-1} \text{ yr}^{-1}$. A steady-state level was reached about 15 years after pasture establishment. These results indicated that topsoil C content was slightly higher in the fertilised plots than in unfertilized plots, but increasing the rate of Superphosphate from 188 to 376 kg ha^{-1} gave no further increase.

Longer term equilibrium soil C on the Winchmore long-term fertiliser trial show no effect of fertiliser treatment on soil C levels (Stewart and Metherell, 1999a; Metherell, 2003), despite the large differences in herbage production. This, in part, results from the greater proportional allocation of C to plant roots in the absence of superphosphate so that there is much less difference in root production between treatments (Stewart and Metherell, 1999b; Metherell, 2003). Plant material with a high lignin content and a low N concentration in the unfertilised treatment (Metherell, 2003) will decompose more slowly. Differences in the quality of plant herbage and roots arise from both the direct effects of nutrient availability and the influence of nutrient availability on botanical composition. The decomposition rate would also be slower because of the lower earthworm biomass when superphosphate was not applied (Fraser *et al*, 1994).

There is also little difference between the soil C levels on the grazed and irrigated trial site and the ungrazed, unirrigated, unfertilized ‘wilderness area’.

Future plans:

Nil. There should be reanalysis of the complete historical series by one analytical method to resolve some discrepancies between the results presented by Nguyen and Goh (1990) and more recent analyses of the 0-7.5 cm samples.

2.9 LONG-TERM (1975–2007) FERTILIZER AND SHEEP GRAZING EXPERIMENTAL SITE: BALLANTRAE

SUMMARY

Two 10 ha farmlets, one having low (LF) and the other high (HF) fertilizer input, were established in 1975 on the Ballantrae Hill Country Research Station of the then DSIR and more latterly AgResearch. The LF farmlet received an average of 125 kg superphosphate (SSP)/ha/yr. The HF farmlet received an average of 625 kg SSP/ha/yr from 1975 to 1979, as well as 1250 kg lime in 1975 and 2500 kg lime in 1979. Since 1980 one HF farmlet has received 375 kg SSP/ha/yr, while the other nil. Since 1980 one LF farmlet has received 125 kg SSP/ha/yr, while the other nil. The initial Olsen P levels of both farmlets was 5 $\mu\text{P/g}$ soil and the retention was low (21-34%). The pH of the LF farmlet was 5.1 and 5.4 for the HF farmlet. Both farmlets were grazed with set stocked Romney breeding ewes. The stocking rate was initially 6 ewes/ha (1974) and this was increased in subsequent years in accordance with changes in pasture production.

Average annual pasture production from 1980-87 was 12.9 kg DM ha⁻¹ for the HF farmlet and 8.4 t DM ha⁻¹ for the LL farmlet. Withholding fertilizer from the HF system resulted in a reasonably consistent decrease in pasture production of 4.6% p.a. from 1980-87. The decline in pasture production from withholding fertilizer from the low input system was much more erratic, but was, on average, 1.7% p.a. over the same period. Withholding fertilizer had little effect on the botanical composition of the pastures or the seasonality of pasture production. The performance of the farmlets receiving no fertilizer has continued to decline, with reversion a major issue on the LF

The experimental site has been used in a large number of studies exploring the interaction between pastoral agriculture and the environment over the last 30 years. Changes in soil C to a depth of 75 mm across these four farmlets and a number of other systems at Ballantrae was published by Lambert et al., (2000). These four farmlets were sampled to two depths 0-75 and 75-150 mm, with separate BD measurements in 2004 at a total of 72 sites. The sites covered 3 slopes and aspects. A preliminary analysis has been completed.

DATABASE DESCRIPTION

Name of the dataset:	Long-term fertilizer and sheep grazing experiment (Ballantrae)
Primary contact:	Alec Mackay
Data ownership / accessibility:	AgResearch
Data storage:	Data is currently held in an Excel spreadsheet
Broad Land-use, Landform sampled:	Set stock sheep pasture Hill land
Number of sites:	72 sites
Geographical spread:	AgResearch Hill Country Research Station, Ballantrae, Manawatu region, New Zealand (40°18'S 175°50'E). Located 300 m above sea level, average air temperature of 12°C and annual rainfall of 1270 mm)
Soil orders:	Brown and Pallic
Funding:	FRST programmes: FRST funded: Bioindicator (1995-1998) and SLURI (2006-current).
Depth of sampling: 0-30 cm:	No

Specific sampling depths:	0-7.5 and 7.5-15 cm
Bulk Density measurements:	Yes
Multiple sampling through time:	No
Pre 1990 soil sampling:	Yes
Site information available	Yes. Very detailed sheep stocking rates, fertilizer inputs etc.
Information type:	GPS coordinates
Soil description	Yes
Are soil samples archived?	Yes
Total change in C or rate of change in soil C recorded:	Yes
Land use history recorded:	Yes
If yes above then history for what period?	1972-2007. land use/management recorded at the sampling times (about 10 years ago)
Management history available	All sites have been set stocked with sheep since 1972 (35 years).
History timeframe:	See above
Management factors held:	Good record of fertilizer inputs, stocking rates, pasture production, etc
Data held in what form:	Excel spreadsheets

Mitigation opportunities to minimize or improve soil C: Nil

Supplementary information on soils e.g. %total N, pH, production:

Total N, Olsen P, total P and S, microbial biomass-C, N P soil physical, earthworms, meso- and macro-fauna, N and S and cation leaching losses, Cd levels

Associated publications:

- Mackay, A.D., Lambert and Barker D.J. 1999. Effect of intensification of livestock farming on the physical properties of a hill soil. In: Best soil management practices for production (eds. L.D. Currie et al.). Occasional Report No. 12, Fertiliser and Lime Research Centre, Massey University, Palmerston North, pp. 311–316.
- Lambert, M.G., Clark, D. A., Mackay, A. D., Costall, D. A. 2000. Effects of fertiliser application on nutrient status and organic matter content of hill soils. N.Z. J. Agric. Sci. 43. 127–138.
- Parfitt, R.L., Yeates, G.W., Ross, D.J., Mackay, A.D., Budding, P.J 2004. Relationships between soil biota, nitrogen and phosphorus availability and pasture growth under organic and conventional management. Applied Soil Ecology 28 1–13
- Barker, D.J., Lambert, M.G., Springett, J.A., Mackay, A.D. 2007. Root characteristics of hill pastures following 22 years of contrasting inputs and sheep stocking rates. Agric. Ecosystems Environ. 43: (Submitted)

Free-form database description

Origin:

This data set was collected in 2004 to quantify the long-term changes in the chemistry and biology of hill soils under low and high fertilizer and sheep grazing pressures. Soil C was one of several measures. The purpose of this dataset was to document the changes that had occurred since the late 1987, when the data was last summarized and published (Lambert et al., 2000).

Collection methodology:

A full description of the sampling approach is not available at this stage. In brief permanent monitoring sites were established on each farmlet back in 1975. Three slope classes and three aspect each replicated twice are located within each farmlet. Each of these sites was sampled to two depths in 2004. Separate cores were collected for chemical analysis and bulk density.

Sample analysis:

Total C, N, P and S were measured on the air-dried samples using a LECO furnace. Soil fertility was assessed by the standard procedures.

Main findings relevant to change in soil C stocks:

Further analysis is required.

Future plans:

To complete the analysis and write up.

2.10 LONG-TERM COMPARISON OF CONVENTIONAL AND ORGANIC SHEEP AND BEEF PRODUCTION

SUMMARY

A long-term replicated farm systems study (1997-2007) examined changes in the biology of mixed-livestock systems associated with the shift to organic production is reported. Two farmlets were managed using conventional farm practices (Con) and the two organic (Org) farmlets complied with the organic production standards of BIO-GRO New Zealand.

This study represents a world first; a long-term replicated farm systems study examining the changes in the biology of legume-based, mixed-livestock systems associated with the shift to organic production.

Soil and pasture sampling have been limited. The opportunity exists to complete a comprehensive comparison of the changes in soil C under conventional and organic practices under very controlled experimental conditions.

DATABASE DESCRIPTION

Name of the dataset:	Organic/conventional comparison
Primary contact:	Alec Mackay
Data ownership / accessibility:	AgResearch
Data storage:	Data is currently held in an Excel spreadsheet and Access Database
Broad Land-use, Landform sampled:	Pasture (mainly dairy and sheep/beef), Hill and flat land
Number of sites:	60 sites
Geographical spread:	AgResearch Hill Country Research Station, Ballantrae, Manawatu region, New Zealand (40°18'S 175°50'E). Located 300 m above sea level, average air temperature of 12°C and annual rainfall of 1270 mm.)
Soil orders:	Brown and Pallic
Funding:	FRST programmes: FRST-funded: Natural and Organic
Depth of sampling: 0-30 cm:	No
Specific sampling depths:	0-7.5 cm
Bulk Density measurements:	Yes
Multiple sampling through time:	Yes
Pre 1990 soil sampling:	No
Site information available	Yes. Very detailed sheep stocking rates, fertilizer inputs etc.
Information type:	GPS coordinates
Soil description	Yes
Are soil samples archived?	No
Total change in C or rate of change in soil C recorded:	No
Land use history recorded:	Yes
If yes above then history for what period?	1988–2007. Land use/management recorded at the sampling times (about 10 years ago)

Management history available	Organic and conventional comparison.
History timeframe:	See above
Management factors held:	Good record of fertilizer inputs, stocking rates, pasture production, etc
Data held in what form:	Excel spreadsheets

Mitigation opportunities to minimize or improve soil C: Nil

Supplementary information on soils e.g. %total N, pH, production:

Total N, Olsen P, total P and S, microbial biomass-C, N P soil physical, earthworms, meso and macro-fauna, N and S and cation leaching losses, Cd levels.

Associated publications:

Parfitt, R.L., Yeates, G.W., Ross, D.J., Mackay, A.D., Budding, P.J 2004. Relationships between soil biota, nitrogen and phosphorus availability and pasture growth under organic and conventional management. *Applied Soil Ecology* 28 1-13

Mackay A.D. Devantier B.P. Pomroy, W.E. 2006. Long-term changes in the biology of a livestock farm system associated with the shift to organic supply. *N Z Grsslds. Assoc.* 68. (133-137)

Free-form database description

Description of farmlets:

Each farmlet is self contained and a scaled model (1:10) of a commercial farm operation. Livestock remain in the farmlet through the study period. The farmlets are located at AgResearch “Ballantrae” Hill Country Research Station in the foothills of the Ruahine Ranges (Lat. Long. 175°50’ E, 40°19’ S) (Mackay *et al.* 1991). The climate at “Ballantrae” is mild-temperate. Mean monthly soil temperatures at 10 cm depth range between 16.0° and 7.2°C and mean annual rainfall of 1276 mm is distributed evenly throughout the year (Lambert *et al.* 1983).

The farmlets are approximately 18 ha in size (range 17.5-18.9 ha) and have similar topography (moderate to steep hill country) aspects (north-east to north-west) and soil fertility (Olsen P <12-20 µg P/cm³ soil). Each farmlet is fenced into 18-20 paddocks (Fig.1). The Con and Org farmlets are fertilised annually with the same amount of reactive phosphate rock (RPR) and elemental sulphur. Low-fertility grasses, such as browntop (*Agrostis capillaris*) and sweet vernal (*Anthoxanthum odoratum*) are the dominant (37 to 79%) grass species. Perennial ryegrass (*Lolium perenne*) makes up a small part of the sward (5–8%) and legumes, including white clover (*Trifolium repens*), 2–21%.

The Organic farmlets complied with the organic production standards of BIO-GRO New Zealand (BIO-GRO New Zealand 2001), an organic certifier and organic producers’ organization in New Zealand. The standards prohibit the routine use of drenches, vaccines, antibiotics, dips and other chemical remedies unless an individual animal suffers or shows signs of ill thrift. One of the Organic farmlets has been registered with BIO-GRO since 1988 and the other since 1997.

Collection methodology:

Soil sampling has been limited. The farmlets represent a resource for exploring the influence of organic practices on soil organic matter and biology

Main findings relevant to change in soil C stocks:

The two farm systems have been compared for the last 10 years under controlled conditions. There is little evidence to suggest that the systems are either diverging or converging with respect to animal production or animal health. The limited sampling of the biology of the soils, constraints any commentary on possible changes in the biology of the soils under the contrasting farm systems.

Future plans:

The farmlets will be disestablished in the spring of 2008. There are no plans at this stage to sample the farmlets.

2.11. AGRESEARCH SOIL BIOINDICATOR DATASET

SUMMARY

From approximately 1997-2001, Landcare Research conducted a program to sample and analyse a large number of soils from the predominant New Zealand intensive agronomic land uses (dairy pasture, sheep and beef pasture, cropping and horticulture, plantation forestry and indigenous vegetation) and encompassing all the major soil Orders across New Zealand. The project was titled the 500 soils project as it was felt a target of sampling and analysing 500 soils would be needed to accomplish this task. A strict series of sampling protocols were used (the 500 soils sampling protocol) and soils were collected from 0-10 cm depth (along a 50 m transect) at each site. It was planned that sites would be re-sampled on a regular basis (varying between 3-10 yrs depending upon land use) to monitor temporal trends in soil quality indicators. Sampling by individual regional councils has continued to the present on a commercial basis with Landcare Research.

DATABASE DESCRIPTION

Name of the dataset:	500 Soils Dataset
Primary contact:	Bryan Stevenson
Data ownership / accessibility:	Landcare Research
Data storage:	Data is currently held in an Excel spreadsheet soon to be updated to database format
Broad Land-use, Landform sampled:	All land uses (dairy, sheep/beef, crop, plantation forestry, indigenous)
Number of sites:	Currently 500+ sites
Geographical spread:	New Zealand-wide (although more concentrated in N. Island)
Soil orders:	Allophanic, Gley, Brown, Pumice, Recent, Ultic
Funding:	FRST/MFE for 500 soils project, commercial (through RCs) after 2001
Depth of sampling: 0-30 cm:	No
Specific sampling depths:	0-10 cm
Bulk Density measurements:	Yes
Multiple sampling through time:	Yes (variable from 3-5+ yrs)
Pre 1990 soil sampling:	No
Site information available	Yes
Information type:	GPS coordinates (Map coordinates for earliest sites)
Soil description	Yes
Are soil samples archived?	Yes (for most sites)
Total change in C or rate of change in soil C recorded:	No (unless remeasured more than once)
Land use history recorded:	Sometimes (sketchy in detail however)
If yes above then history for what period?	Variable
Management history available	Variable
History timeframe:	See above
Management factors held:	Fertilizer when available
Data held in what form:	Excel spreadsheets

Mitigation opportunities to minimise or improve soil C:

Moderate. After several samplings (ideally 3-5) trend analysis will be performed to determine changing rates of soil quality indicators (including total C) on different landuses

Supplementary information on soils e.g. %total N, pH, production:

pH, Total N, Olsen P, anaerobically mineralisable N, bulk density, macroporosity.

Associated publications:

Selected publications (does not include individual reports to regional councils)

- Giltrap DJ, Hewitt AE 2004. Spatial variability of soil quality indicators in New Zealand. soils and land uses. *New Zealand Journal of Agricultural Research* 47: 167–177.
- Hill RB, Sparling G, Frampton C, Cuff J 2003. National soil quality review and programme design: Environmental reporting. In: *Signposts for sustainability*. Wellington, Ministry for the Environment. Technical Paper No 75, Land. Pp.1–34.
- Sparling G P, Schipper LA 2004. Soil quality monitoring in New Zealand: Trends and issues arising from a broad-scale survey. *Agriculture Ecosystem & Environment* 104: 545–552.
- Sparling G P, Schipper LA, Bettjeman W, Hill R 2004. Soil quality monitoring in New Zealand: Practical lessons from a six-year trial. *Agriculture Ecosystems & Environment* 104: 523–534.
- Sparling GP, Lilburn L, Vojvodic-Vucovic M 2003. Provisional targets for soil quality indicators in New Zealand. Landcare Research, Palmerston North. 60 p.
- Sparling G, Schipper L, McLeod M, Basher L, Rijkse W 1996. Trailing soil quality indicators for State of the Environment monitoring: Research report for 1995/1996. Unpublished Landcare Research Contract Report LC9596/149. Hamilton, Landcare Research.
- Sparling GP, Rijkse W, Wilde RH, van der Weerden T, Beare MH, Francis GS 2001a. Implementing soil quality indicators for land. Research Report for 2000/2001 and Final Report for MfE Project Number 5089. Unpublished Landcare Research Contract Report LC0102/015. Hamilton, Landcare Research.
- Sparling GP, Rijkse W, Wilde RH, van der Weerden T, Beare MH, Francis GS 2001b. Implementing soil quality indicators for land: Research report for 1999/2000. Unpublished Landcare Research Contract

Free-form database description**Origin:**

This data set was originally started as part of the 500 soils program funded by MFE, FRST and Regional councils. The dataset has grown to 700+ records (more to be added), however, records added the end of the 500 soils program are strictly the IP of RC's. Through an Envirolink grant, we are in the process of obtaining funding to incorporate the entire dataset into a database format and finalise IP clauses for the entire dataset.

Collection methodology:

Transect sampling

Standardised methodology employed:

Yes – see Hill et al. 2003.

Sample analysis:

All the soil biochemical properties were determined using field moist samples. Total C, N and S were measured in the air-dried samples using a LECO furnace.

Main findings relevant to change in soil C stocks:

None to this point

Future plans:

Sampling continues on a regional basis (largely North Island RCs).

2.12 LANDCARE RESEARCH 500+ SOIL DATASET

SUMMARY

From approximately 1997-2001, Landcare Research conducted a program to sample and analyse a large number of soils from the predominant New Zealand intensive agronomic landuses (dairy pasture, sheep and beef pasture, cropping and horticulture, plantation forestry and indigenous vegetation) and encompassing all the major soil Orders across New Zealand. The project was titled the 500 soils project as it was felt a target of sampling and analysing 500 soils would be needed to accomplish this task. A strict series of sampling protocols were used (the 500 soils sampling protocol) and soils were collected from 0-10 cm depth (along a 50 m transect) at each site. It was planned that sites would be re-sampled on a regular basis (varying between 3-10 yrs depending upon landuse) to monitor temporal trends in soil quality indicators. Sampling by individual regional councils has continued to the present on a commercial basis with Landcare Research.

DATABASE DESCRIPTION

Name of the dataset:	500 Soils Dataset
Primary contact:	Bryan Stevenson
Data ownership / accessibility:	Landcare Research
Data storage:	Data is currently held in an Excel spreadsheet soon to be updated to database format
Broad Land-use, Landform sampled:	All land uses (dairy, sheep/beef, crop, plantation forestry, indigenous)
Number of sites:	Currently 500+ sites
Geographical spread:	New Zealand wide (although more concentrated in N. Island)
Soil orders:	Allophanic, Gley, Brown, Pumice, Recent, Ultic
Funding:	FRST/MFE for 500 soils project, commercial (through RCs) after 2001
Depth of sampling: 0-30 cm:	No
Specific sampling depths:	0-10 cm
Bulk Density measurements:	Yes
Multiple sampling through time:	Yes (variable from 3-5+ yrs)
Pre 1990 soil sampling:	No
Site information available	Yes
Information type:	GPS coordinates (Map coordinates for earliest sites)
Soil description	Yes
Are soil samples archived?	Yes (for most sites)
Total change in C or rate of change in soil C recorded:	No (unless remeasured more than once)
Land use history recorded:	Sometimes (sketchy in detail however)
If yes above then history for what period?	Variable
Management history available	Variable
History timeframe:	See above
Management factors held:	Fertilizer when available
Data held in what form:	Excel spreadsheets

Mitigation opportunities to minimise or improve soil C:

Moderate. After several samplings (ideally 3-5) trend analysis will be performed to determine changing rates of soil quality indicators (including total C) on different landuses.

Supplementary information on soils:

pH, Total N, Olsen P, anaerobically mineralisable N, bulk density, macroporosity.

Associated publications:

A few selected publications, does not include individual reports to Regional councils.

Giltrap DJ, Hewitt AE 2004. Spatial variability of soil quality indicators in New Zealand. soils and land uses. *New Zealand Journal of Agricultural Research* 47: 167–177.

Hill RB, Sparling G, Frampton C, Cuff J 2003. National soil quality review and programme design: Environmental reporting. In: *Signposts for sustainability*. Wellington, Ministry for the Environment. Technical Paper No 75, Land. Pp.1–34.

Sparling G P, Schipper LA 2004. Soil quality monitoring in New Zealand: Trends and issues arising from a broad-scale survey. *Agriculture Ecosystem & Environment* 104: 545–552.

Sparling G P, Schipper LA, Bettjeman W, Hill R 2004. Soil quality monitoring in New Zealand: Practical lessons from a six-year trial. *Agriculture Ecosystems & Environment* 104: 523–534.

Sparling GP, Lilburn L, Vojvodic-Vucovic M 2003. Provisional targets for soil quality indicators in New Zealand. Landcare Research, Palmerston North. 60 p.

Sparling G, Schipper L, McLeod M, Basher L, Rijkse W 1996. Trailing soil quality indicators for State of the Environment monitoring: Research report for 1995/1996. Unpublished Landcare Research Contract Report LC9596/149. Hamilton, Landcare Research.

Sparling GP, Rijkse W, Wilde RH, van der Weerden T, Beare MH, Francis GS 2001a. Implementing soil quality indicators for land. Research Report for 2000/2001 and Final Report for MfE Project Number 5089. Unpublished Landcare Research Contract Report LC0102/015. Hamilton, Landcare Research.

Sparling GP, Rijkse W, Wilde RH, van der Weerden T, Beare MH, Francis GS 2001b. Implementing soil quality indicators for land: Research report for 1999/2000. Unpublished Landcare Research Contract

Free-form database description**Origin:**

This data set was originally started as part of the 500 soils program funded by MFE, FRST and Regional councils. The dataset has grown to 700+ records (more to be added), however, records added the end of the 500 soils program are strictly the IP of RC's. Through an Envirolink grant, we are in the process of obtaining funding to incorporate the entire dataset into a database format and finalise IP clauses for the entire dataset.

Collection methodology:

Transect sampling

Standardised methodology employed:

Yes (See Hill et al. 2003)

Sample analysis:

All the soil biochemical properties were determined using field moist samples. Total C, N and S were measured in the air-dried samples using a LECO furnace.

Main findings relevant to change in soil C stocks:

None to this point

Future plans:

Sampling continues on a regional basis (largely North Island RCs).

APPENDIX 3: DATASETS—Cropping and Soil Carbon

3.1 LAND MANAGEMENT INDEX DATASET

SUMMARY

The Land Management Index (LMI) dataset was collected for the purpose of developing a decision support system that farmers and land managers can use to (1) track changes in soil quality and predict risks to productivity losses or gains based on current management, and (2) to predict the effects of a change-in-management on soil quality and productivity before applying the change to the paddock.

The dataset comprises soil quality indicator measurements from 746 paddocks sampled between July 2002 and July 2007 as part of the LMI project. The paddocks represent seven land uses (mixed and intensive arable and vegetable cropping, dairy pasture, intensive bull/beef pasture and extensive sheep/beef pasture) spread across seven different regions (Canterbury, Southland, Auckland, Waikato, Hawke's Bay, Manawatu and Gisborne) in New Zealand. The paddocks sampled are located on key soil types representative of the major agricultural land uses in each region. The soil carbon (C) data were collected from 0-15 and 15-30 cm sample depths and are accompanied by bulk density measurements at these same depths. The LMI soil quality dataset is closely aligned to comprehensive soil and crop management history information that is held in Crop & Food Research's Soil and Land Management Database. The database contains detailed information on the management practices used to establish and manage the crops and pastures (tillage types & frequency, irrigation, fertiliser, crop residue management, grazing practices) grown during the 10 years preceding the measurement of LMI indicators. The primary contacts for the data are Mike Beare & Erin Lawrence at Crop and Food Research, Lincoln.

The LMI dataset was not collected for the explicit purpose of soil carbon accounting. However, the dataset represents what is probably the largest and most comprehensive datasets suitable for quantifying soil C stocks under the major agricultural land uses in New Zealand. The dataset can also be used to quantify the magnitude of soil C stock change under land use change and the impacts of specific management factors (e.g. irrigation, tillage, winter cover crops) on C stocks in key land uses. The LMI dataset does not include data for many horticultural land uses (e.g. Pipfruit, kiwifruit, viticulture etc), as well as forestry and hill and high country pastoral farming systems. Some soil orders and several regions (e.g. Gisbone, Manawatu) are also underrepresented in the dataset. The dataset also lacks C stock data for several other regions in New Zealand (e.g. Northland, Taranaki, Wanganui), although all of the major cropping regions are represented.

DATABASE DESCRIPTION

Name of the dataset	Land Management Index (LMI)
Primary contact:	Dr Mike Beare or Ms Erin Lawrence, New Zealand Institute for Crop & Food Research (CFR)
Data ownership:	Joint ownership – Crop & Food Research, SFF, HortNZ, FAR, Auckland Regional Council, Environment Waikato, Hawke’s Bay Regional Council, Horizons Manawatu, Environment Canterbury, Environment Southland. The Soil and Land Management Database is owned by Crop & Food Research.
Accessibility:	By approval of joint owners, normally in collaboration with Crop & Food Scientists
Data storage:	Data held in Excel spreadsheets and CFR Soil & Land Management Database (a Microsoft Access Database).
Broad Land-use/Land form sampled:	Intensive and mixed arable and vegetable cropping, dairy pasture, intensive bull/beef pasture, and extensive sheep/beef pasture.
Number of sites:	746 paddocks sampled.
Geographical spread:	North and South Island.
Soil orders:	Allophanic (127), Brown (128), Gley (142), Granular (58), Melanic (6), Organic (8), Pallic (168), Recent (105), Ultic (1), Unidentified (3).
Depth of sampling: 0-30 cm	Yes
Specific sampling depths:	0-15 cm, 15-30 cm.
Bulk Density measurements:	Yes
Multiple sampling through time:	No
Pre 1990 soil sampling:	No
Site location information available:	Yes
Information type:	GPS
Soil description:	No
Are soil samples archived?	Yes
Total or rate of change in soil C recorded:	The dataset allows quantification of soil C stocks under major agricultural land uses across New Zealand and the impacts of specific management factors (e.g. irrigation, tillage, winter cover crops) on C stocks in key land uses.
Land use history recorded:	Yes
If Yes, history for what period?	10 years preceding soil sampling.
Management history available:	Yes
History timeframe:	Monthly to yearly time steps for 10 yrs preceding sampling.
Management factors:	Tillage type, crop type plus sowing and harvest dates (where available), crop yields, stocking rate (some), irrigation type, fertiliser type/rate (some), residue management (some).
Data format:	Microsoft access database and Excel spreadsheets.

Mitigation opportunities to minimise or improve soil C:

The primary purpose of the LMI dataset was to develop an on-farm soil and crop management decision support system that predicts the effects of changing management practices on soil quality and future productivity. This includes the quantification of soil C stocks under the primary agricultural land uses across New Zealand and the impacts of specific management factors (e.g. irrigation, tillage, winter cover crops) on C stocks in each major soil orders. Consequently, when fully validated, the model will allow us to compare the effects of specific management factors on soil C storage and loss and to identify management systems that have the greatest potential to maintain or improve soil C stocks.

Supplementary information on soils:

Indicator	Units	0-15 cm	15-30 cm
Total C and N	% , t/ha	Y	Y
C:N	Ratio	Y	Y
Hot-water extractable carbon	µg/g	Y	-
Aggregate stability	MWD (mm), %<1 mm	Y	-
Aggregate size distribution	MWD (mm), %<0.85 mm, %>9.5 mm	Y	-
Bulk density	g/cm ³	Y	Y
Penetration resistance	MPa (moisture corrected)	Y	Y
Olsen P	µg/g	Y	-
pH		Y	-
Soil texture	Hand texture analysis	Y	-

Associated publications:

- Lawrence, E.J., M.H. Beare, C.S. Tregurtha, and J. Cuff 2008 Quantifying the effects of soil and crop management history on soil quality. *In:* (Eds L.D. Currie and J.A. Hanly) Occasional Report. Fertilizer & Lime Research Centre, Massey University, Palmerston North (In press).
- Lawrence, EJ, MH Beare, CS Tregurtha, J Cuff 2008. The Land Management Index. Presented at the LandWISE Annual Conference 2008, Advanced Farming Systems. 14 & 15 May 2008, Gisborne.
- Beare, M., E. Lawrence, C Tregurtha, and J Cuff. 2007. Land Management Index: a soil management decision support tool. Poster presented to Hort NZ annual conference, Christchurch, 1-2 August 2007.
- Beare, M., E. Lawrence, C Tregurtha, T. Harrison-Kirk and J Cuff. 2007. Progress toward development of the Land Management Index – June 2007, CFR confidential report no 1839. 64 p.
- Beare, M.H., E.J. Lawrence, C.S. Tregurtha, T. Harrison-Kirk, A. Pearson, and E.D. Meenken. 2005. Progress in development of the Land Management Index: 2004-05 project report. Crop & Food Research Confidential Report No. 1408. Crop & Food Research, 46 p.

Free-form database description

Origin:

The LMI dataset was collected for the primary purpose of developing a decision support tool designed to help farmers understand how soil and crop management affects soil quality and future productivity.

The LMI model development and the underpinning soil quality dataset were funded by the Sustainable Farming Fund, the Foundation for Arable Research, Horticulture New Zealand, six different regional councils (Canterbury, Southland, Waikato, Auckland, Manawatu, and Hawke's Bay) and Crop & Food Research.

Collection methodology:

The paddocks were selected to represent the dominant agricultural land uses within key soil orders in each of the seven participating regions. Where possible, the soil types included for sampling were those most representative of each land use in a given region. The paddocks selection criteria were refined further to allow comparison of specific management practices (crop type, tillage type & frequency, winter grazing strategies) to be investigated for potential effects on soil quality and productivity.

The seasonal timing of the on-farm measurements and sample collection differed by land use. Paddocks under cropping land uses were sampled immediately prior to or shortly after crop harvest but before cultivation and sowing of the next crop. In general, most of the paddocks under dairy production were sampled in the spring while those under intensive bull/beef and extensive sheep/beef farming were sampled in the autumn. Care was taken to avoid sampling under very wet or very dry conditions wherever possible.

Composite soil samples were collected from each three replicate sample locations in each paddock. The composite samples from the surface soils (0-15 cm) were composed of three 7.2 cm diameter cores. A single 15-30 cm composite sample was collected from each paddock, which was composed of two soil cores taken from each of the three sample locations. The samples were kept in sealed plastic bags and chilled until processing for laboratory analysis. Full collection methodology, sample preparation and analysis techniques are detailed in Beare et al. (2007).

Sample analysis relevant to soil C:

Total C concentration was determined on a LECO CNS200 furnace/gas analyser (McGill & Figueirido 1993) after oven drying soil (60°C) overnight.

Main findings relevant to change in soil C stocks:

Preliminary analysis of the LMI dataset has been completed for the purposes of this report, within the confines of the available time and funding.

The individual soil C concentrations and bulk density measurements were used to calculate the soil C stocks (t C ha^{-1}) in the top 30 cm of soil at each sample location in each sampled paddock. The paddock average values were then used to calculate the average C stocks for each land use by soil order combination. A detailed analysis of land use effects for all soil orders represented in the dataset and the individual soil types is beyond the scope of this project. The data presented in Figures 3.1 and 3.2 are preliminary only and require closer scrutiny before being adopted for policy applications.

Key points:

- 1) The median C stocks tend to be highest under dairy pasture and lowest under intensive vegetable cropping regardless of soil order.
- 2) In general, Brown and Pallic soils tend to have narrower range of values than Gley and Allophanic soils
- 3) The effects of different cropping land uses on soil C stocks relative to sheep/beef pasture tend to be greater for Allophanic and Gley soils than for Brown and Pallic soils.

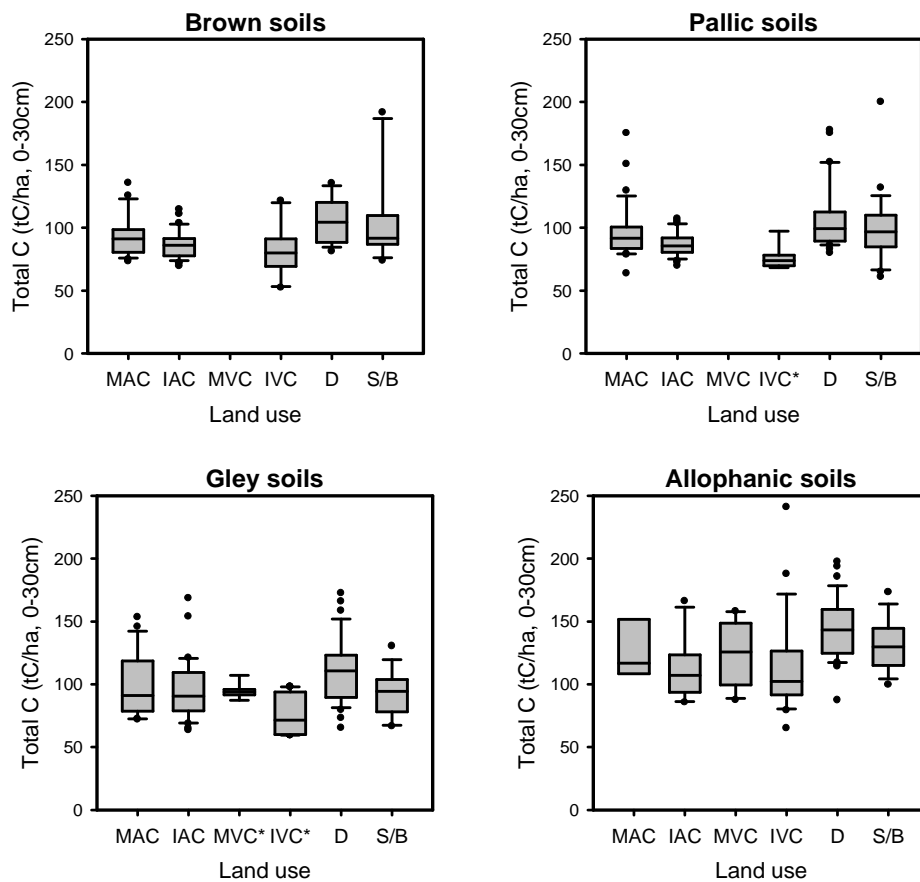


Figure 3.1. Box and whisker plots showing the distribution of soil C stocks representing each of the major land uses sampled under four major soil orders (Brown, Pallic, Gley and Allophanic soils). The land uses plotted are: mixed arable cropping (MAC), intensive arable cropping (IAC), mixed vegetable cropping (MVC), intensive vegetable cropping (IVC), dairy (D) and extensive sheep/beef pasture (S/B). Each box represents the middle 50% of the values measured for each land use, the line across each box is the median value and the values plotted outside the boxes are the upper and lower quartiles of values.

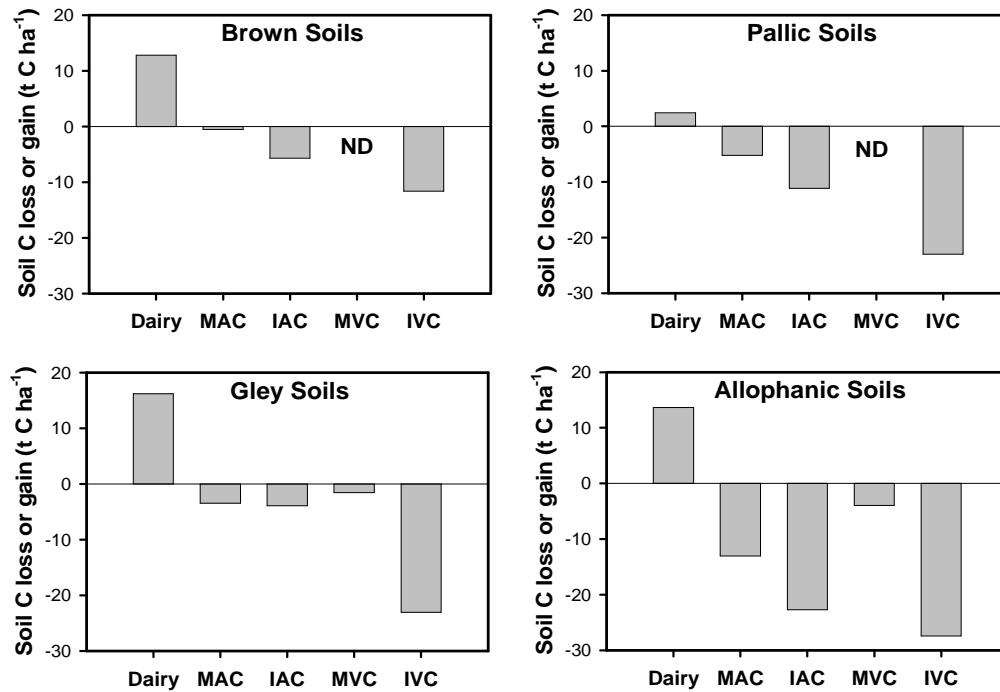


Figure 3.2. The soil C loss or gain for each major land use relative to the C stock under extensive sheep pasture for four major soil order (Brown, Pallic, Gley and Allophanic soils). The values plotted are based on the difference between the median values for each land use by soil order combination. Negative values represent C loss and positive values represent C gains.

Future plans:

The LMI dataset was not collected for the explicit purpose of soil carbon accounting. However, the dataset represents what is probably the largest and most comprehensive dataset suitable for quantifying soil C stocks under the major agricultural land uses in New Zealand. The dataset can also be used to quantify the magnitude of soil C stock change under land use change and the impacts of specific management factors (e.g. irrigation, tillage, winter cover crops) on C stocks in key land uses. Preliminary findings from this dataset are described above. A much more comprehensive analysis of the data set could provide valuable information on:

- 1) The mean, median and statistical range of soil C stocks in the top 30 cm of the soil profile under the major agricultural land uses, by soil order and dominant soil types.
- 2) The magnitude of C stock change under different agricultural land uses relative to extensive sheep pasture.
- 3) The effects of key management factors under different cropping (mixed and intensive arable and vegetable cropping practices in various regions) and pastoral land uses (Dairy, extensive sheep and intensive bull/beef farming in key regions) on soil C stock change.
- 4) The management systems that offer the greatest potential of mitigating C losses.

3.2 ECAN A&P DATASET

SUMMARY

In 1999, Environment Canterbury (ECan) and Crop & Food Research (CFR) initiated a long-term soil quality monitoring programme to obtain information on soil conditions for different land use/soil type combinations. The Arable and Pastoral Monitoring dataset forms part of ECan's state of the environment monitoring and reporting programme.

As of June 2007 the Arable and Pastoral Monitoring dataset comprised soil quality indicator data from 220 paddocks sampled between 1999 and 2006. The paddocks sampled represent three broad land use categories (i.e. long-term pasture, short term pasture or arable and long-term arable cropping) on each of 12 different soil types commonly found on the Canterbury plains and downs. From 2008 onward, previously sampled sites are scheduled for re-sampling (9 years post original sampling) in order to provide information on changes in soil quality over time. The soil C data collected prior to June 2006 was based on 0-15 cm samples only, while data collected since that time and in the future will be based on the top 30 cm (0-15 and 15-30 cm) of the soil profile. The soil C data are accompanied by bulk density measurements and a number of other soil chemical, physical and biological measurements. The primary contacts for the data are Mike Beare & Craig Tregurtha at Crop and Food Research, Lincoln.

The Arable and Pastoral Monitoring dataset was not collected for the explicit purpose of soil carbon accounting. The current dataset has limited application for C accounting owing to the shallow depth (0-15 cm) of sampling. However the existing data does provide a relative measure of the resistance of different soil types to C loss following conversion from long-term pasture to continuous cropping land uses. From 2008 onwards, the C stock measurements made to 30 cm will have more direct applications to soil C accounting and quantifying the change in soil C stocks from long-term pasture to long-term continuous cropping land uses on different soil types.

DATABASE DESCRIPTION

Name of the dataset	ECan Arable & Pastoral Monitoring.
Primary contact:	Dr Mike Beare or Mr Craig Tregurtha, New Zealand Institute for Crop & Food Research
Data ownership:	Joint ownership - Crop & Food Research and Environment Canterbury
Accessibility:	By approval of joint owners, normally in collaboration with Crop & Food Scientists
Data storage:	Data held in Excel spreadsheets, some complimentary management history data held in the Soil and Land Management Database (Microsoft Access). Soil quality data also held by Environment Canterbury.
Broad Land-use/Land form sampled:	Long-term pasture, short term pasture or arable and long-term arable cropping
Number of sites:	220 paddocks
Geographical spread:	Canterbury plains and downs
Soil orders:	Brown (67), Gley (23), Pallic (114), Recent (16).
Depth of sampling: 0-30 cm	No (1999 – 2006); Yes (2007 to present)
Specific sampling depths:	1999 – 2006 (0-15 cm), 2007 to present (0-15 cm, 15-30 cm).

Bulk Density measurements:	Yes
Multiple sampling through time:	No [Yes from 2008 onwards]
Pre 1990 soil sampling:	No
Site location information available:	Yes
Information Type:	GPS
Soil description:	No
Are soil samples archived?	pre 2003 (No), post 2003 (yes)
Total or rate of change in soil C recorded:	The dataset allows quantification of C stocks under all three land uses for each major soil type and a means of estimating the magnitude of potential change from one steady state condition to another.
Land use history recorded:	Yes
If Yes, history for what period?	10 years preceding soil sampling.
Management history available:	Yes (Crop type by year, more detail from 2008 onward).
History timeframe:	10 yrs preceding sampling.
Management factors:	Crop type sown. From 2008 data will include tillage type, crop type plus sowing and harvest dates (where available), crop yields, stocking rate, irrigation type, fertiliser type/rate, residue management, etc.
Data format:	Microsoft Excel spreadsheets.

Mitigation opportunities to minimise or improve soil C:

The dataset can be used to evaluate the resistance of different soil types to soil C loss and soil quality change under change land use from long-term pasture to long-term continuous cropping in Canterbury.

Supplementary information on soils:

Indicator	Units	0-15	15-30
		cm	cm
Total C and N	%, g/cm ³ , t/ha	Y	Y ¹
C:N ratio	Ratio	Y	Y ¹
Hot-water extractable carbon	µg/g	Y ¹	-
Aggregate stability	MWD (mm), %<1 mm	Y	-
Aggregate size distribution	MWD (mm), %<0.85 mm, %>9.5 mm	Y ¹	-
Bulk density	g/cm ³	Y	Y ¹
Penetration resistance	MPa (moisture corrected)	Y ¹	Y ¹
Olsen P	µg/g	Y ¹	-
pH		Y ¹	-
Soil texture	Hand texture analysis	Y ¹	-
Anaerobically mineralisable N	µg/g, µg/cm ³	Y	-
Cadmium	µg/g, µg/cm ³	Y	-

¹ Data available from 2007 onwards.

Associated publications:

Lawrence EJ, Francis GS, Beare MH, Tregurtha CS. 2006. Regional environmental monitoring programme for soil quality 2005-2006. Crop and Food Research Confidential Report No. 1644. Christchurch, New Zealand Institute for crop & Food Research Ltd. 116 p.

Tregurtha CS, Lawrence EJ, Beare MH. 2007. Regional environmental monitoring programme for soil quality 2006-2007: Arable and Pastoral Project. Crop and Food Research Confidential Report No. 1987. Christchurch, New Zealand Institute for crop & Food Research Ltd. 42 p.

Beare MH, EJ Lawrence and CS Tregurtha 2007. Soil C in cropping soils, New Zealand Soil Carbon and Climate Change Workshop. Ministry for Agriculture and Forestry. 23 November, 2007. Wellington.

Free-form database description**Origin:**

The Arable and Pastoral Monitoring dataset has been collected to meet Environment Canterbury's requirement for state-of-environment reporting, and to obtain information on the resistance of different soil types to land use change.

The collection, analysis and interpretation of the dataset has been funded entirely by Environment Canterbury.

Collection methodology:

The paddocks sampled represent three broad land use categories (i.e. long-term pasture, short term pasture or arable and long-term arable cropping) on each of 12 different soil types commonly found on the Canterbury plains and downs. Strict criteria were applied to the selection of these paddocks. The soil types selected for monitoring included those that cover a relatively large area (>20,000 ha) of the Canterbury plains and are commonly used for mixed and intensive arable cropping and extensive sheep/beef farming.

The on-farm measurements and sample collection were completed in the autumn of each year. In most cases the paddocks were sampled immediately prior to or shortly after crop harvest but before any cultivation for autumn sowing. Care was taken to avoid sampling under very wet or very dry conditions wherever possible.

Separate soil samples were collected from each of three or four replicate sample locations per paddock. The composite samples from each location were composed of three soil cores (7.2 cm diameter) taken from the top 15 cm of the soil. Soil samples were kept in sealed plastic bags and chilled until processing for laboratory analysis. Full collection methodology, sample preparation and analysis techniques are detailed in Lawrence et al. (2006).

Sample analysis relevant to soil C:

Total C concentration was determined on a LECO CNS200 furnace/gas analyser (McGill & Figueirido 1993) after oven drying soil (60°C) overnight.

Main findings relevant to change in soil C stocks:

Preliminary analysis of the Arable and Pastoral monitoring dataset has been completed for the purposes of this report, within the confines of the available time and funding.

The individual soil C concentrations and bulk density measurements were used to calculate the soil C stocks (t C ha^{-1}) in the top 15 cm of soil at each sample location in each sampled paddock. The paddock average values were then used to calculate the average C stocks for each land use by soil order combination. A detailed analysis of the land use effects for the individual soil types is beyond the scope of this project but could be included in future MAF funded soil C stocks and change research.

The results showed that the magnitude of soil C stock change from long-term pasture to long term arable cropping differs significantly between soil orders (Figure 3.3). In general the lighter, well drained Brown soils had the lowest C stocks under longer-term pasture (LT Pasture) and smallest loss of soil C under long-term continuous cropping (LT Arable) relative to long-term pasture. In contrast, the heavier, usually poorly drained Gley soils tended to maintain the highest C stocks under long-term pasture and showed the greatest losses of C under long-term continuous cropping. The soil C stocks under cropping on Pallic soils (mostly imperfectly drained) tended to be lower than those of the Brown and Gley soils, though the losses of C relative to long-term pasture were intermediate to those of the Brown and Gley soils.

Future plans:

ECan is committed to repeat sampling of each paddock in the programme (on a 9 year return cycle). With suitable funding, the C stock measurements made from 2008 onward (top 30 cm) will have useful applications to soil C accounting and quantifying the change in soil C stocks from continuous pasture to continuous cropping land uses on different soil types.

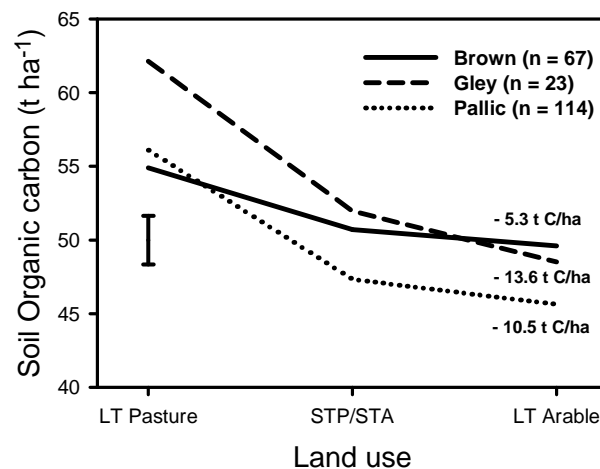


Figure 3.3. Average soil C stocks (t C ha^{-1}) in the top 15 cm of Brown, Gley and Pallic soils under long-term sheep pasture (LT Pasture), short-term pasture of arable cropping (STP/STA) and long-term continuous arable cropping (LT Arable). The difference in C stocks between LT Pasture and LT Arable is also shown as the average C loss under continuous cropping, n = the number of paddocks representing each soil order in the data set.

3.3 LAND USE CHANGE AND INTENSIFICATION (LUCI) – CANTERBURY

SUMMARY

The goal of the FRST-funded Land Use Change and Intensification (LUCI) programme (2003-2008) is to provide integrated knowledge and tools required by land users and policy makers to assess the environmental impacts associated with land use change and intensification of agricultural practices. One key focus of this programme has been to quantifying the extent and rate of change in soil quality and plant production following changes in land use under typical management practices in Canterbury.

The extent and rate of soil quality change is being quantified under several important forms of land use change (LUC) on the Canterbury plains. This is based on replicated paddocks undergoing conversion from extensive dryland sheep pasture or dryland mixed cropping to intensive irrigated cropping on soil types representing well drained, imperfectly drained and poorly drained soils. The dataset includes annual measurement of soil physical, chemical and biological properties on paddocks undergoing the first four to five years of land use change and on other paddocks representing longer periods of intensive irrigated cropping (up to 20 yrs) on the same soil types. The soil types included in this analysis of land use change impacts were selected to represent well drained, imperfectly drained and poorly drained soils common to the Canterbury Plains. As of June 2007, a total of 87 paddocks had been sampled, with many of these paddocks involved in repeated measurement of the soil quality indicators. The soil C data were collected from 0-15 and 15-30 cm sample depths and are accompanied by bulk density measurements at these same depths. The primary contacts for the data are Mike Beare & Erin Lawrence at Crop and Food Research, Lincoln.

The LUC dataset was not collected for the explicit purpose of soil carbon accounting. However, it represents one of only a few datasets suitable for quantifying the actual rate of soil C change under land use change in New Zealand. The dataset allows quantification of soil C stocks ($t\ C\ ha^{-1}$, 0-30 cm) under major agricultural land uses on the Canterbury and the impacts of specific management factors (e.g. irrigation, tillage, winter cover crops) on C stocks in key land uses. The LUC soil quality dataset is closely aligned to comprehensive soil and crop management history information that is held in Crop & Food Research's Soil and Land Management Database. The database contains detailed information on the management practices used to establish and manage the crops and pastures (tillage types & frequency, irrigation, fertiliser, crop residue management, grazing practices) grown during the 10 years preceding the soil quality measurements.

Collection of LUC soil quality dataset will be complete in June 2008. Detailed analysis of the data is scheduled to follow. The dataset allows quantification of C change, both rate and extent of C change with change in management.

Closely aligned and complimentary data on the rate of soil quality (including soil C) change during the conversion of extensive dryland sheep pasture or dryland mixed cropping to intensive irrigated dairy farming is also being collected for well drained soils in Canterbury under the Sustainable Land Use Research Initiative (SLURI).

DATABASE DESCRIPTION

Name of the dataset	Land Use Change and Intensification (LUCI)
Primary contact:	Dr Mike Beare, New Zealand Institute for Crop & Food Research
Data ownership:	Crop & Food Research
Accessibility:	By approval of Crop & Food Research, normally in collaboration with their scientists
Data storage:	Data held in Excel spreadsheets, some management history data held in the Soil and Land Management Database in association with the LMI programme
Broad Land-use/Land form sampled:	Dryland sheep pasture, mixed cropping and irrigated intensive cropping.
Number of sites:	87 sites (but many of these sites had repeat samplings)
Geographical spread:	Canterbury plains
Soil orders:	Brown (32), Gley (27), Pallic (28)
Depth of sampling: 0-30 cm	Yes
Specific sampling depths:	0-15 cm, 15-30 cm
Bulk Density measurements:	Yes
Multiple sampling through time:	Yes
Pre 1990 soil sampling:	No
Site location information available:	Yes
Information Type:	GPS coordinates
Soil description:	No
Are soil samples archived?	Yes
Total or rate of change in soil C recorded:	Yes
Land use history recorded:	Yes
If Yes, history for what period?	Land use history for the 20 years proceeding the sampling
Management history available:	Yes
History timeframe:	Year-by-year data for 10 yrs proceeding sampling, and general land use history information for the period between 10-20 years before sampling.
Management factors:	Tillage type, crop type plus sowing and harvest dates (where available), crop yields, stocking rate, irrigation type, fertiliser type/rate, residue management.
Data format:	Excel spreadsheets and CFR's Soil and Crop Management access database.

Mitigation opportunities to minimise or improve soil C:

Dataset can be used to identify which soils are most resistant to soil C loss under specific forms of land use change in Canterbury and to quantify actual rates of soil C loss or gain under the forms of land use change. The dataset will also provide information on the effects of short term irrigation on soil C.

Supplementary information on soils:

Measures	Units	0-7.5 cm	0-10 cm	0-15 cm	15-30 cm	By horizon to 1.5 m	All paddocks
DUL	%, v/v	X				x	
Kq						x	
DLL	%, v/v					x	
Depth to stones	cm					x	
Stone content	%					x	
Texture	Hand texture			X	x		
Bulk density	g/cm ³			X	x	x	
Soil moisture	%			X	x	x	
Total porosity	%			X	x	x	
Total C&N	%			X	x		
C&N mineralisation	mg/kg soil/d			X	x		
SCS	1-10 score		X				
HWEC	µg/g			X			
Crop Yield	t/ha						x
Aggregate size distribution	MWD (mm), %		X				
GPS							x

Associated publications:

Beare, M. 2008 Measurement and management of soil carbon in cropping soils. Invited presentation to the annual LandWISE conference. Advance Farming Systems. 14-15 May, Gisborne.

Free-form database description**Origin:**

The LUC dataset was collected as part of the FRST funded programme called Land Use Change and Intensification. The programme goal is to provide integrated knowledge and tools required by land users and policy makers to assess the environmental impacts associated with changes in, and intensification, of land use. One key focus of this programme has been to quantifying the extent and rate of change in soil quality and plant production following changes in land use under typical management practices in Canterbury.

Collection methodology:

Three soil types representing well drained (Lismore), imperfectly drained (Wakanui) and poorly drained (Temuka) soils were sampled. For each soil type the following paddocks were sampled:

1. four paddocks under the steady state pre-conversion land use of dryland pasture,
2. four paddocks in year one of conversion from dryland pasture to irrigated intensive cropping, with each of these paddocks measured annually to collected data for 1, 2, 3 and 4 post-conversion,
3. four paddocks that had been converted from dryland pasture to irrigated intensive cropping between 8 and 10 years prior to sampling, and
4. four paddocks that had been converted from dryland pasture to irrigated intensive cropping more than 15 years prior to sampling (assumed to be at steady state).

The same criteria and sampling schedules were applied to paddocks representing the previous land use of dryland mixed cropping. The paddocks sampled under number 4 above were repeat sampled 3 years later to confirm their assumed steady state conditions.

The on-farm measurements and sample collection were completed in the autumn of each year. In most cases the paddocks were sampled immediately prior to or shortly after crop harvest but before any cultivation for autumn sowing. Care was taken to avoid sampling under very wet or very dry conditions wherever possible.

Composite samples for soil C analysis were collected from each of three replicate sample locations in each paddock. The composite samples from the surface soils (0-15 cm) were composed of three 7.2 cm diameter cores. A single 15-30 cm composite sample was collected from each paddock, which was composed of two soil cores taken from each of the three sample locations. The samples were kept in sealed plastic bags and chilled until processing for lab analysis. Full collection methodology, sample preparation and analysis techniques are available upon request.

Sample analysis relevant to soil C:

Total C concentration was determined on a LECO CNS200 furnace/gas analyser (McGill & Figueirido 1993) after oven drying soil (60°C) overnight.

Main findings relevant to change in soil C stocks:

The collection of this dataset has only recently been completed and it awaits detailed analysis to determine the main findings.

Future plans:

Analysis and interpretation of the data will focus on quantifying the extent and rate of change in soil quality and plant production following changes in land use under typical management practices in Canterbury. Although the data set was not explicitly collected for purposes of soil C accounting, this objective could be included as part of the data analyses planned for 2008/09. Some supplementary funding would be required to assist with this analysis and the data interpretations.

3.4 SOIL QUALITY MANAGEMENT SYSTEM (SQMS) DATASET

SUMMARY

This SQMS dataset was collected as part of the development of the Soil Quality Management System (SQMS). The SQMS was developed to assist farmers with on-farm monitoring of soil quality to improve soil management decisions. Soil quality indicator data (including soil C measurements) were collected from mixed and intensive cropping and extensive sheep paddocks representing some of the most common cropping soils in the Canterbury and Southland regions.

The Canterbury dataset includes annual measurements of soil quality on 69 paddocks between 1999 and 2001. The Southland paddocks (31) were each sampled annually between 2002 and 2004. The soil C data are based on composite samples made up of 15 soil cores collected along a W or Z transect in each paddock. The soil C data are accompanied by bulk density measurements and a number of other soil chemical, physical and biological measurements. Soil and crop management history information (crop type, tillage type, irrigation, residue management, etc) was also collected for the 10 years preceding the soil quality assessments in each paddock. The primary contacts for the data are Mike Beare & Craig Tregurtha at Crop and Food Research, Lincoln.

The SQMS dataset was not collected for the explicit purpose of soil C accounting. The dataset has limited application for this purpose owing to the shallow depth (0-15 cm) of sampling. However the existing data does provide a measure of the range of topsoil C stocks for extensive sheep and cropping land uses in Canterbury. When combined with the management history information, the dataset has also proved valuable in evaluating the effects of key management factors (e.g. tillage type and intensity, residue management practices, crop rotations etc) on topsoil quality. A further analysis of the dataset is needed to quantify the impacts of management practices on soil C stocks.

DATABASE DESCRIPTION

Name of the dataset	Soil Quality Management System (SQMS)
Primary contact:	Dr Mike Beare, New Zealand Institute for Crop & Food Research
Data ownership:	Crop & Food Research
Accessibility:	By approval from Crop & Food Research, normally in collaboration with Crop & Food Scientists
Data storage:	Soils data held in Excel spreadsheets. Complimentary management history data (tillage type/frequency, crop type, residue management, irrigation) held in separate Excel spreadsheets.
Broad Land-use/Land form sampled:	Mixed and intensive arable cropping, and extensive sheep/beef pasture.
Number of sites:	Canterbury – 69 paddocks Southland – 31 paddocks
Geographical spread:	Canterbury and Southland
Soil orders:	Canterbury - Brown (24), Gley (7), Pallic (26), Recent (2)

	Southland - Brown (10), Gley (6), Pallic (13), Recent (2)
Depth of sampling: 0-30 cm	No
Specific sampling depths:	0-15 cm
Bulk Density measurements:	Yes
Multiple sampling through time:	Canterbury - Yes (1999, 2000, 2001) Southland - Yes (2002, 2003, 2004)
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	GPS (Longitude/latitude coordinates) & farm/paddocks maps
Soil description:	No
Are soil samples archived?	Yes
Total or rate of change in soil C recorded:	Magnitude of change can be assessed by comparing soil C stocks under long-term (steady state) arable and sheep pasture paddocks. The rate of C loss and gain in mixed cropping rotations can be quantified using the management history information.
Land use history recorded:	Yes
If Yes, history for what period?	10 years preceding soil sampling
Management history available:	Yes
History timeframe:	10 yrs proceeding sampling.
Management factors:	For each crop sown in sequence - tillage type, crop residue management practice, irrigation type, grazing, etc. Crop yield data is available for many paddocks, but normally only for the crop immediately prior to sampling.
Data format:	Microsoft Excel spreadsheets.

Mitigation opportunities to minimise or improve soil C:

The dataset was collected to describe the range of soil quality conditions found on arable cropping farms, at different stages of arable cropping rotations and under different management practices. The dataset can be used to investigate the optimum length of mixed cropping rotation to maintain C stocks within a specified target range and to evaluate the effects of primary management factors (e.g. tillage type and intensity, residue management, crop rotations, etc) on soil C stocks in the top 15 cm of the soil profile.

Supplementary information on soils:

<i>Indicator</i>	<i>Units</i>	<i>Measurement Depth</i>
Total C and N	%, t/ha	0-15 cm
Olsen P	µg/g	0-15 cm
Exchangeable K, Ca, Mg	µg/g	0-15 cm
pH		0-15 cm
Aggregate stability	MWD (mm), %<1 mm	0-7.5 cm
Bulk density	g/cm ³	0-7.5 cm
Penetration resistance	Mpa (moisture corrected)	0-20 cm
Structural condition score	1-10 score	0-10 cm
Profile density score	8-24 score	0-15 cm & subsoil
Earthworm	Number/ m ²	0-25 cm

Associated publications:

Beare, M.H. and C.S. Tregurtha 2004. Soil Quality on Southland Cropping Farms: A guide to monitoring and best management practices. New Zealand Institute for Crop & Food Research, Christchurch, ISBN 0478108486, 51 p.

Beare, M.H., C.S. Tregurtha and E.J. Lawrence. 2003. Quantifying the effects of management history on soil quality and crop performance using the soil quality management system. Pp 167-174. In: *Tools for Nutrient and Pollutant Management: applications to agriculture and the environmental quality*. (Eds L.D. Currie and J.A. Hanly) Occasional Report No. 17. Fertilizer & Lime Research Centre, Massey University, Palmerston North.

Free-form database description**Origin:**

The SQMS dataset was collected to describe the range of soil quality conditions (including soil C concentrations and stocks) found on arable cropping farms in Canterbury and Southland, and provide information to underpin the development of the SQMS on-farm monitoring and management system. The dataset was not collected with the explicit purpose of quantify changes in soil C stocks for C accounting purposes.

The development of SQMS in Canterbury and Southland and collection of the underpinning datasets were funded in part by Crop & Food Research, the Sustainable Farming Fund, AgMARDT, the Foundation for Arable Research.

Collection methodology:

The farms included in the dataset were selected to represent a wide range of mixed and intensive cropping practices on most of the dominant soil types used for cropping in Canterbury and Southland. The paddocks selected for monitoring on each farm were chosen to represent different stages of the crop rotations practiced on each farm.

The on-farm measurements and sample collection were done in autumn of each year. In most cases the paddocks were sampled immediately prior to, or shortly after, crop harvest but before any cultivation for autumn sowing. Care was taken to avoid sampling under very wet or very dry conditions wherever possible.

The soil C measurements were made from a composite sample composed of 15 soil cores (2.5 cm diam x 15 cm deep) collected along a W or Z transect in each paddock. The composite samples were kept in sealed plastic bags and chilled prior to processing for laboratory analysis. The samples were sieved (<2 mm) field moist and a sub-sample removed for air-drying prior to C analyses. Full collection methodology, sample preparation and analysis techniques are detailed in Beare and Tregurtha (2004).

Sample analysis relevant to soil C:

Total C concentration was determined on a LECO CNS200 furnace/gas analyser (McGill & Figueirido 1993) after oven drying soil (60°C) overnight.

Main findings relevant to change in soil C stocks:

The dataset has not been adequately analysed with respect to soil C stocks and change.

Future plans:

Parts of the SQMS soil quality dataset are currently being written up for publication in international journal articles. Analysis of the top soil C stocks data in relation to management history may be included in this analysis.

3.5 MILLENNIUM TILLAGE TRIAL (MTT) DATASET

SUMMARY

The Millennium Tillage Trial (MTT) was established to identify tillage and cover crop management practices that maintain organic matter levels, reduce structural degradation, increase nutrient use efficiency and minimize nutrient losses in order to sustain arable cropping out of an improved condition under long-term grass pasture. The MTT is part of the FRST funded Land Use Change and Intensification (LUCI) programme (2003-2008) and contributes to the development of best management practices for the intensification of arable cropping. However, the dataset also has the potential to contribute to our understanding of the short-term soil quality changes (including change in C stocks) that occurs during establishment and maintenance of break crops in pastoral re-grassing (grass renewal) rotations.

The trial is composed of six tillage treatments (based on different combinations of spring & autumn tillage) plus a 'control' of uncultivated permanent pasture alongside a permanent fallow treatment. The three main tillage methods used were Intensive tillage, minimum tillage and no-tillage. Each of the main plots has been split to compare winter cover crops to no cover crop (winter fallow) treatments. Each treatment is replicated three times, giving a total of 42 treatment plots. Details on the trial design, including the specific management practices used can be found in the Cropping Section.

The MTT dataset includes annual measurements of a wide range of soil physical, chemical and biological properties from the treatment plots on Lincoln based trial site. The soil at this site is Wakanui silt loam that had been under at least 15 years of continuous sheep pasture management prior to establishing the trial. The soil C measurements have been made annually from composite samples made up seven soil cores (each 7.2 cm in diameter) collected from 0-7.5, 7.5-15, 15-25 and 25-30 cm sample depths in each plot. The soil coring equipment was designed to collect relatively large samples suitable for bulk density analyses and a wide range of other physical, chemical, biological measurements at each depth. Considerable care has been taken to complete very precise accounting of soil C and N stocks under the different treatments over time using the equivalent mass sampling method (Ellert et al. 2001). The primary contacts for the data are Mike Beare, Denis Curtin and Trish Fraser at Crop and Food Research, Lincoln.

The MTT dataset was not collected for the explicit purpose of soil carbon accounting. However, it represents one of only a very few datasets that are suitable for quantifying the extent and rate of soil C change during the conversion of pasture to arable cropping in New Zealand. It also represents the only long-term replicated trial whereby the effects of different tillage practices (including no-tillage) on soil C stocks can be evaluated. The dataset currently contains annual measurements of soil C stocks (t C ha^{-1} , 0-30 cm) under each of six different tillage systems (ranging from no-tillage to continuous intensive tillage) in an arable cropping rotation that was established out of grass pasture and has been under continuous treatment for 8 years. We are currently reviewing the trial to determine if it should be maintained in the longer-term.

DATABASE DESCRIPTION

Name of the dataset	Millennium Tillage Trial (MTT)
Primary contact(s):	Drs Mike Beare and Denis Curtin, New Zealand Institute for Crop & Food Research
Data ownership:	Crop & Food Research
Accessibility:	By approval from Crop & Food Research, normally in collaboration with Crop & Food Scientists
Data storage:	Soils data is held in Excel spreadsheets. Complimentary information on the management inputs (e.g. irrigation, fertiliser, herbicides etc) to the trial and regular soil moisture and temperature monitoring data are also held in separate Excel spreadsheets.
Broad Land-use/Land form sampled:	Intensive arable cropping (under different tillage practices), extensive sheep pasture and chemical fallow treatments.
Number of sites:	1 site, 14 treatments, each replicated 3 times (42 plots)
Geographical spread:	Lincoln
Soil orders:	Pallic (Wakanui silt loam)
Depth of sampling: 0-30 cm	Yes
Specific sampling depths:	0-7.5 cm, 7.5-15 cm, 15-25 cm, 25-30 cm
Bulk Density measurements:	Yes
Multiple sampling through time:	Yes (annually 2000-2008)
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	GPS & farm map
Soil description:	Yes
Are soil samples archived?	Yes
Total or rate of change in soil C recorded:	Magnitude and rate of C stock change can be described for the different tillage and cover crop management practices maintained in the trial
Land use history recorded:	Yes
If Yes, history for what period?	14 years preceding establishment of the trial
Management history available:	Yes
History timeframe:	For each year of the trial (2000-2008)
Management factors:	For each crop sown in sequence - tillage type, crop residue management practice, rainfall and irrigation, winter grazing, fertiliser, herbicide and pesticide inputs,

etc.. Detailed crop performance and dry matter production data is also available for all treatments in all years.

Data format:

Microsoft Excel spreadsheets.

Supplementary information on soils:

<i>Indicator</i>	<i>Units</i>	<i>Measurement Depth</i>
Total C and N	%, t/ha, equivalent mass	0-7.5, 7.5-15, 15-25, 25-30 cm
Olsen P	µg/g	0-7.5, 7.5-15, 15-25 cm
Exchangeable K, Ca, Mg	µg/g	0-7.5, 7.5-15, 15-25 cm
pH		0-7.5, 7.5-15, 15-25 cm
Microbial biomass C & N	µg/g, t/ha	0-7.5, 7.5-15, 15-25 cm
Hot water extractable C	µg/g, t/ha	0-7.5, 7.5-15, 15-25 cm
Coarse organic matter C & N	µg/g, t/ha	0-7.5, 7.5-15, 15-25 cm
Particulate organic matter C & N	µg/g, t/ha	0-7.5, 7.5-15, 15-25 cm
Particle size fraction C & N ¹	µg/g, t/ha	0-7.5, 7.5-15, 15-25 cm
Light fraction C & N ¹	µg/g, t/ha	0-7.5, 7.5-15, 15-25 cm
Anaerobically mineralisable N	µg/g, t/ha	0-7.5, 7.5-15, 15-25 cm
Aerobically mineralisable C & N ¹	µg/g, t/ha	0-7.5, 7.5-15, 15-25 cm
Bulk density	g/cm ³	0-7.5, 7.5-15, 15-25, 25-30 cm
Soil structural condition score	Score (1-10)	0-10 cm
Aggregate stability	MWD (mm)	0-7.5, 7.5-15, 15-25 cm
Aggregate size distribution	MWD (mm)	0-7.5, 7.5-15, 15-25 cm
Pore size distribution	% (v/v)	0-7.5, 7.5-15, 15-25 cm
Nitrate leaching	Kg NO ₃ -N leached	60 cm
Particle size fraction C & N ¹	µg/g, t/ha	0-7.5, 7.5-15, 15-25 cm
Light fraction C & N ¹	µg/g, t/ha	0-7.5, 7.5-15, 15-25 cm
Earthworm populations ²	Number/ m ² , biomass	0-25 cm
Slug populations	Number/ m ²	Soil surface
Soil moisture (TDR)		0-20 and 0-60 cm
Soil Temperature	°C	0-7.5 cm
Plant production data	Yield, dry matter, emergence etc.	All plots, each year

¹ Available on selected sample dates

² Late winter / early spring populations

Associated publications:

Beare, MH, PF Fraser and DC Curtin (2008). Trial design, management and main crop yields, The Foundation for Arable Research, Arable Extra No. 77. 4 p.

3.6 STRAW FIELD TRIAL DATASET

As post-harvest crop residues represent a major input of C to arable cropping systems, the way in which these residues are managed may have a significant influence on soil C stocks. Effects of three straw management practices [(1) straw incorporated; (2) straw baled and removed, and (3) straw burned] on soil C were determined in a six-year study (1992-98) in Canterbury (Lincoln) (Curtin and Fraser, 2003).

In the straw-incorporated treatment, about 25 t/ha of straw (~11 t C/ha) was returned to the soil during the course of the trial. However, there was no significant effect ($P > 0.05$) of straw management on total soil C (0-15 cm). Measurements of straw decomposition using the litter bag technique (carried out in association with the trial) indicated that much of the incorporated straw would have decomposed and the small fraction of straw-C retained in the soil (estimated at 2-3 t C/ha) would have been difficult to detect against a background soil C content of over 50 t/ha in the top 15 cm.

Limitations: Duration of trial was not sufficiently long. Estimates of C gains due to crop residue retention range from 0.1 t/ha.year (IPCC, 2000) to 0.7 t/ha.year in some European studies (Smith et al., 2005). Ideally trial should be 15-20 years to detect small annual gains in soil C.

Sampling was to 15 cm.

Table 3.1. Effect of straw management on soil C after 6 years.

Treatment	0-7.5 cm		7.5-15 cm		0-15 cm
	C conc. (g/kg)	BD (g/cm ³)	C conc. (g/kg)	BD (g/cm ³)	C mass (t/ha)
Incorporated	31.5	1.04	31.6	1.19	52.8
Burned	31.3	1.19	30.9	1.21	55.9
Removed	31.3	1.08	31.2	1.17	52.7

3.7 ORGANICS DATASETS

Published work

Formal trials providing direct comparison of C stocks under organic and conventional arable cropping have not been conducted in New Zealand. A number of published studies are available in which paired comparisons were made between commercial organic farms and nearby conventional farms (Reganold et al., 1993; Nguyen et al., 1995; Murata and Goh, 1997). These studies have been reviewed by Condon et al. (2000) who concluded that soil organic matter is generally higher under organic vs. conventional production systems.

The Reganold study, which included a wide range of land uses (market gardens, citrus orchards, mixed farms, livestock farms, dairy farms) in the North Island, compared properties that had been under a biodynamic regime for at least 8 years (the range was 8 to 18 years) with adjacent conventional farms. Carbon concentration (samples taken to 10 cm) was higher for the biodynamic system in five out of seven comparisons. The mean C concentration (aggregated across land uses) was significantly higher under the biodynamic regime (4.84 vs. 4.27% C; $P < 0.01$). Mean bulk density (only data for the top 5 cm was presented) was significantly lower in biodynamic farms (1.07 vs. 1.15 Mg/m³), so that differences in C stocks between the two systems would be less than suggested by the C concentration values. Assuming that the presented bulk density values are applicable to the entire 0-10 cm layer, the estimated mass of C in the top 10 cm averages 51.8 t/ha for the biodynamic farms compared with 49.1 t/ha for the conventional farms. This dataset has the serious limitation that sampling was restricted to the top 10 cm and it would be unwise to conclude from this study that biodynamic management enhances C storage.

In Canterbury (Rakaia), Murata and Goh (1997) sampled a biodynamic (8 years after conversion to biodynamic) and an adjacent conventional farm, both of which followed a mixed cropping rotation (arable cropping rotated with short-term pasture). Samples were collected from both the cropping and arable phases of the rotation (0-7.5 and 7.5-15 cm depths).

On the conventional farm, seven cropping and three pasture paddocks were sampled while on the biodynamic farm four cropping and four pasture paddocks were sampled. The mean C concentration in the 0-7.5 cm layer tended to be higher in the biodynamic farm (Table 3.2), with the conventional farm tending to be higher in C in the 7.5-15 cm depth. As bulk density values were not reported, it is not possible to compare C stocks in the organic and conventional systems. An assumption implicit in this, and other, studies using the paired-farm approach to compare organic and conventional management is that C levels were the same at the time of conversion to the organic system.

Nguyen et al. (1995) sampled paddocks in the cropped and pastoral phases on conventional and alternative (biodynamic or organic) mixed cropping farms in Canterbury. The alternative systems had been in place for 7-8 years prior to sampling. Three pairs of farms were sampled, covering three soil types (Kowai sandy loam, Temuka silt loam, Templeton silt loam). Samples were taken from the 0-7.5 and 7.5-15 cm depths, but C concentrations were reported for the 0-7.5 cm layer only. Bulk density values were not reported.

Soil C concentration was higher under alternative than conventional pastoral management in one comparison (Kowai soil) (Table 3.3). Under arable cropping, there was no difference between alternative and conventional system on any of the three soil types.

This dataset is of limited use because (1) data are for the top 7.5 cm only and (2) bulk density values are no available.

Table 3.2. Mean C concentration (%) in the 0-7.5 and 7.5-15 cm soil depths in the cropping and pasture phases of a mixed cropping rotation in adjacent biodynamic and conventional farms in Canterbury (adapted from Murata and Goh, 1997).

Management	Cropping phase		Pasture phase	
	0-7.5 cm	7.5-15 cm	0-7.5 cm	7.5-15 cm
Conventional	3.2	3.2	3.3	3.1
Biodynamic	3.6	3.0	3.6	3.0

Table 3.3. C concentrations (%; 0-7.5 cm) in cropped and pastoral soils under conventional and organic management (adapted from Nguyen et al., 1995).

Management	Kowai		Temuka		Templeton	
	Cropped	Pastoral	Cropped	Pastoral	Cropped	Pastoral
Conventional	3.4	3.5	3.4	3.4	2.9	3.1
Alternative ¹	3.4	3.9	3.3	3.5	3.0	3.2

¹The alternative system was “biodynamic” for the Kowai farms and “organic” for the other two.

Kowhai Farm data set (unpublished)

SUMMARY

Kowhai farm was established as a joint venture between Lincoln University and Heinz-Watties in spring 1999 to demonstrate the economic viability and environmental sustainability of farm-scale certified organic production. This demonstration farm is composed of 6 paddocks, most of which are represented by Wakanui, Templeton and Paparua silt loam soils. This dataset was collected to describe changes in soil quality during the conversion to certified organic production.

The indicators used for monitoring include those recommended as part of the Soil Quality Management System (SQMS). These include measurements of soil C (0-15 cm only) and a wide range of other chemical, physical and biological indicators. The SQMS indicators were also measured on several of the Kowhai Farm paddocks prior to initiating the organic farm conversion and on other adjoining conventional cropping paddocks prior to, and during, conversion to fully certified organic production. Soil and crop management history information (crop type, tillage type, irrigation, residue management, etc) was also collected for a period of about 10 years preceding the soil quality assessments on each paddock.

The Kowhai Farm SQMS dataset was not collected for the explicit purpose of soil C accounting. The dataset has limited applications for C accounting owing to the shallow depth (0-15 cm) of sampling. However the existing data does provide a measure of the potential

change in topsoil C stocks during conversion from conventional to certified organic cropping on free draining soil of Canterbury. Interpretation of these data in relation to the effects of organic production on soil C stocks will require a more comprehensive review of management history information available for this site.

DATABASE DESCRIPTION

Name of the dataset	Kowhai Farm – SQMS
Primary contact:	Dr Mike Beare, New Zealand Institute for Crop & Food Research
Data ownership:	Crop & Food Research
Accessibility:	By approval from Crop & Food Research, normally in collaboration with Crop & Food Scientists
Data storage:	Soils data held in Excel spreadsheets. Some complimentary management history data (e.g. tillage type/frequency, crop type, residue management, fertiliser) is held in separate Excel spreadsheets.
Broad Land-use/Land form sampled:	Mixed arable cropping under conversion to certified organic production.
Number of sites:	6 paddocks
Geographical spread:	Lincoln University cropping farm
Soil orders:	Pallic
Depth of sampling: 0-30 cm	No
Specific sampling depths:	0-15 cm
Bulk Density measurements:	Yes
Multiple sampling through time:	Yes Prior to conversion (3 paddocks, each year, 1997-1999) During conversion (6 paddocks, each year, 2000-2008) Up to three adjoining conventional cropping farm paddocks
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	GPS & farm/paddocks maps
Soil description:	Yes (Trevor Webb – Landcare Research)
Are soil samples archived?	Yes
Total or rate of change in soil C recorded:	Magnitude and rate of change can be described for the period of continuous cropping under conversion to organic production and by comparison to long- term cropping and extensive sheep pastures under conventional management on the Lincoln Cropping

Land use history recorded:	Farm.
If Yes, history for what period?	Yes In general for the 10 years preceding each soil sampling.
Management history available:	Yes
History timeframe:	For the years immediately preceding and during conversion to certified organic production.
Management factors:	Crop types sown and fertiliser and tillage practices applied to each crop in the crop sequence.
Data format:	Microsoft Excel spreadsheets.

Mitigation opportunities to minimise or improve soil C:

The dataset provides information on changes on top soil (0-15 cm) C stocks during conversion from long-term conventional arable cropping to certified organic production of arable and vegetable crops.

Supplementary information on soils:

Indicator	Units	Measurement Depth
Total C and N	%, t/ha	0-15 cm
Olsen P	µg/g	0-15 cm
Resin P ¹	µg/g	0-15 cm
Quick Test K, Ca, Mg	Test units	0-15 cm
pH		0-15 cm
Anaerobically mineralisable N ²	µg/g	0-15 cm
Bulk density	g/cm ³	0-7.5 cm
Aggregate stability	%<1 mm	0-7.5 cm
Structural condition score	1-10 score	0-10 cm
Penetration resistance	Mpa (moisture corrected)	0-25cm
Profile density score	8-24 score	Topsoil to subsoil
Earthworm	Number/ m ²	0-25 cm

¹ Resin P was added to the standard suite of SQMS indicators because it is sometimes recommended a better measure of plant available P where rock phosphate is applied.

² Anaerobically mineralisable N was added to provide a measure of plant-available N.

Associated publications:

Horrocks, AJ, CS Tregurtha, MH Beare And PM Fraser 2007. Management History And Soil Quality During The Conversion Of Kowhai Farm To Organic Production. Crop & Food Confidential Report No. 1864. New Zealand Institute For Crop & Food Research Ltd., Christchurch. 34p

Free-form database description

Origin:

Kowhai farm was established as a joint venture between Lincoln University and Heinz-Watties in spring 1999 to demonstrate the economic viability and environmental sustainability of farm-scale certified organic production. The Kowhai Farm (SQMS) dataset was collected to describe soil quality changes during conversion from long-term conventional arable cropping to certified organic production of arable and vegetable crops. The dataset was not collected with the explicit purpose of quantify changes in soil C stocks.

Collection methodology:

The paddocks selected for monitoring in this dataset include the six Kowhai Farm organic conversion paddocks and up to three other conventional arable cropping paddocks located on the adjoining Lincoln Cropping Farm. The SQMS indicators were measured on each of six Kowhai Farm paddocks in each year of conversion to certified production from 2000 to 2006 and again in 2008. Three of the six paddocks were also monitored between 1997 and 1999, prior to initiating conversion to organic production. The three conventional cropping paddocks on the adjoining Lincoln Cropping Farm were also monitored in 2001 and 2006.

The soil C measurements were made from a composite sample composed of 15 soil cores (2.5 cm diam x 15 cm deep) collected along a W or Z transect in each paddock. The composite samples were kept in sealed plastic bags and chilled prior to processing for laboratory analysis. The samples were sieved (<2 mm) field moist and a sub-sample removed for air-drying prior to C analyses. Full collection methodology, sample preparation and analysis techniques are detailed in Beare and Tregurtha (2004) and Horrocks et al (2007)..

Sample analysis relevant to soil C:

Total C concentration was determined on a LECO CNS200 furnace/gas analyser (McGill & Figueirido 1993) after oven drying soil (60°C) overnight.

Main findings relevant to change in soil C stocks:

The dataset has not been adequately analysed with respect to soil C stocks and change.

Future plans:

No current plans to analyse these data specifically for the purposes of assessing stocks and change in soil C.

3.8 RESTORATIVE CROPS TRIAL DATASET

SUMMARY

A six-year (1989–95) experiment was carried out by Francis et al. (1999) to evaluate the ability of a variety of crops to improve the fertility and physical condition of an intensively-cropped, degraded soil (Wakanui silt loam) in Canterbury (Lincoln). Treatments included perennial pastures, annual pastures, and arable crops (Table 3.4). Soil C was determined after 3 and 6 years, but treatment effects were generally not significant until the sixth year (Table 3.5). In year 6, soil C (0-20 cm) ranged from 64 to 77 t/ha. Carbon stocks were least in the annual cultivated treatments and greatest in the perennial treatments. The results confirm the benefits of including a pasture phase in a cropping rotation to increase soil C, but the length of pasture may need to be greater than 3 years to achieve a measurable increase in C stocks. The results also indicate that, among perennial treatments, ryegrass was more effective than white clover in raising soil C and that grazed pasture may maintain higher C levels than mown pasture. An interesting observation from this trial is the higher C level under annual ryegrass that was direct drilled vs. annual ryegrass that was established using conventional cultivation.

Limitations: Sampling was to 20 cm only. Data are for a single soil type and location.

Table 3.4. Treatments applied in the Restorative Crops Trial and their associated management practices (Francis et al. 1999).

Treatment code	Plant species	Type of crop	Crop management
PR	Perennial ryegrass	Perennial	Grazed
PRM	Perennial ryegrass	Perennial	Mown
WC	White clover	Perennial	Grazed
RC	Perennial ryegrass/white clover	Perennial	Grazed
HL	Mixed herb ley ¹	Perennial	Grazed
AR	Perennial ryegrass	Annual	Grazed; re-established each year using conventional tillage
ARDD	Perennial ryegrass	Annual	Grazed; re-established each year using direct drilling
BAR	Barley	Annual	Sown in spring; residues burned
LUP	Lupins	Annual	Sown in spring; residues burned

¹The mixed herb ley consisted of white clover, red clover, chicory, prairie grass, timothy, lucerne, and tall fescue.

Table 3.5. Treatment effects on soil C (0-20 cm) after six years.

Treatment code	Carbon (t/ha)			
	0-5 cm	5-10 cm	10-20 cm	0-20 cm
PR	20.1	19.8	37.5	77.4
PRM	16.5	18.5	36.7	71.7
WC	18.3	17.8	35.4	71.5
RC	19.6	19.4	38.6	77.6
HL	19.3	20.2	37.6	77.1
AR	16.9	16.1	33.7	66.7
ARDD	19.1	19.4	37.4	75.9
BAR	12.2	17.3	34.7	64.2
LUP	15.3	16.9	34.0	66.2

3.9 CROPLAND DATA FROM OTHER DATASETS

Sparling-Schipper published studies

Effects of land use on soil quality (including C levels in top 10 cm) have been reported by Schipper and Sparling (2000) and Sparling and Schipper (2004). Of the 511 sites sampled by Sparling and Schipper (2004), 44 were under “arable cropping” and 17 were under “mixed cropping”. The mean C content was 40.7 t/ha (standard error \pm 2.8) and 37.6 t/ha (SE \pm 1.3) for “arable cropping” and “mixed cropping”, respectively. Values for drystock pastures (142 sites) and dairy pastures (127 sites) were substantially higher than those of the arable sites (50.8 and 66.9 t/ha for drystock and dairy, respectively).

Schipper and Sparling (2000) reported C levels under arable cropping and three other land uses (pasture, plantation forest, indigenous forest) from a study that comprised nine soil great groups in the Auckland, Waikato, and Canterbury regions. Matched sites were selected on the same soil great group (sites were in close proximity; within 0-1 km) and differed only in their land use. Carbon was lower under cropping (20-34 t/ha) than in other land uses (pasture 30.7-141.5 t/ha; indigenous forest 31.8-52.9 t/ha).

Limitation of these data sets: Sampling to 10 cm only.

500 Soils Dataset

SUMMARY

From approximately 1997–2001, Landcare Research conducted a programme to sample and analyse a large number of soils from the predominant New Zealand intensive agronomic land uses (dairy pasture, sheep and beef pasture, cropping and horticulture, plantation forestry and indigenous vegetation) and encompassing all the major soil Orders across New Zealand. The project was titled the 500 soils project as it was felt a target of sampling and analysing 500 soils would be needed to accomplish this task. A strict series of sampling protocols were used (the 500 soils sampling protocol) and soils were collected from 0-10 cm depth (along a 50 m transect) at each site. It was planned that sites would be re-sampled on a regular basis (varying between 3-10 yrs depending upon land use) to monitor temporal trends in soil quality indicators. Sampling (and funding) by individual regional councils has continued to the present with Landcare Research.

DATABASE DESCRIPTION

Name of the dataset	500 Soils Dataset
Primary contact:	Bryan Stevenson, Hamilton
Data ownership:	Landcare Research
Accessibility:	Via Landcare Research
Data storage:	Data is currently held in an Excel spreadsheet soon to be updated to database format
Broad Land-use/Land form sampled:	All land uses (dairy, sheep/beef, crop, plantation forestry, indigenous, scrub, urban, wetland).

	Sites by land use, cropping, pasture 339, horticulture 49, indigenous forest 63, plantation forest 79, scrub 8, tussock grassland 23, urban 1, wetland 1, unidentified (includes repeat sampling) 173.
Number of sites:	Currently 846 records
Geographical spread:	New Zealand wide (although more concentrated in N. Island)
Soil orders:	Allophanic, Gley, Brown, Pumice, Recent, Ultic
Depth of sampling: 0-30 cm	No
Specific sampling depths:	0-10 cm
Bulk Density measurements:	Yes
Multiple sampling through time:	Yes (variable from 3-5+ yrs)
Pre 1990 soil sampling:	No
Site Location Information available:	Yes
Information Type:	GPS coordinates (Map coordinates for earliest sites)
Soil description:	Yes
Are soil samples archived?	Yes (for most sites)
Total or rate of change in soil C recorded:	No (unless re-measured more than once)
Land use history recorded:	Sometimes (sketchy in detail however)
If Yes, history for what period?	Variable
Management history available:	Variable
History timeframe:	See above
Management factors:	Fertilizer when available
Data format:	Excel spreadsheets

Mitigation opportunities to minimise or improve soil C:

After several samplings (ideally 3-5) trend analysis will be performed to determine changing rates of soil quality indicators (including total C) on different land uses.

Supplementary information on soils:

pH, Total N, Olsen P, anaerobically mineralisable N, bulk density, macroporosity.

Associated publications:

A few selected publications (does not include individual reports to regional councils).

Giltrap DJ, Hewitt AE 2004. Spatial variability of soil quality indicators in New Zealand. soils and land uses. *New Zealand Journal of Agricultural Research* 47: 167–177.

Hill RB, Sparling G, Frampton C, Cuff J 2003. National soil quality review and programme design: Environmental reporting. In: Signposts for sustainability. Wellington, Ministry for the Environment. Technical Paper No 75, Land. Pp.1–34.

Sparling G P, Schipper LA 1998. Soil quality monitoring in New Zealand: Concepts, approach and interpretation. Landcare Research Technical Report LCR 9798/060. 41p.

Sparling G P, Schipper LA 2004. Soil quality monitoring in New Zealand: Trends and issues arising from a broad-scale survey. *Agriculture Ecosystem & Environment* 104: 545–552.

- Sparling G P, Schipper LA, Bettjeman W, Hill R 2004. Soil quality monitoring in New Zealand: Practical lessons from a six-year trial. *Agriculture Ecosystems & Environment* 104: 523–534.
- Sparling GP, Lilburn L, Vojvodic-Vucovic M 2003. Provisional targets for soil quality indicators in New Zealand. Landcare Research, Palmerston North. 60 p.
- Sparling G, Schipper L, McLeod M, Basher L, Rijkse W 1996. Trailing soil quality indicators for State of the Environment monitoring: Research report for 1995/1996. Unpublished Landcare Research Contract Report LC9596/149. Hamilton, Landcare Research.
- Sparling GP, Rijkse W, Wilde RH, van der Weerden T, Beare MH, Francis GS 2001a. Implementing soil quality indicators for land. Research Report for 2000/2001 and Final Report for MfE Project Number 5089. Unpublished Landcare Research Contract Report LC0102/015. Hamilton, Landcare Research.
- Sparling GP, Rijkse W, Wilde RH, van der Weerden T, Beare MH, Francis GS 2001b. Implementing soil quality indicators for land: Research report for 1999/2000. Unpublished Landcare Research Contract

Free-form database description

Origin:

This data set was originally started as part of the 500 soils program funded by MFE, FRST and Regional councils. The dataset has grown to 700+ records (more to be added), however, records added the end of the 500 soils program are strictly the IP of RC's. Through an Envirolink grant, we are in the process of obtaining funding to incorporate the entire dataset into a database format and finalise IP clauses for the entire dataset.

Collection methodology:

Transect sampling

Standardised methodology employed:

Yes (See Hill et al. 2003)

Sample analysis:

All the soil biochemical properties were determined using field moist samples. Total C, N and S were measured in the air-dried samples using a LECO furnace.

Main findings relevant to change in soil C stocks:

None to this point

Future plans:

Sampling continues on a regional basis (largely North Island RCs).

Francis & Knight Tillage Trial

Francis and Knight (1993) compared the effects of conventional cultivation (ploughing to 18-20 cm) and no-tillage at two sites in Canterbury. One site, on a Lismore stony silt loam, had been under a ryegrass-clover pasture for 5 years prior to the trial while the second site, on a slow-draining Wakanui silt loam, had been under an arable cropping rotation for 10 years. At the end of the 9-year trial, total C (0-15 cm depth) in the Lismore soil had declined from 54 t/ha under pasture to 45 and 39 t/ha under no-tillage and conventional tillage respectively. At

the Wakanui site, there was no difference between tillage treatments after 9 years (total C to 15 cm was 37 t/ha under no-tillage vs. 36 t/ha under conventional tillage).

Limitations of data set: Sampling to 15 cm only – this is a problem as cultivation was to 18-20 cm, so the full cultivation layer was not sampled.

Aslam et al. – tillage effects on soil C

Aslam et al. (1999, 2000) conducted a short-term (two years) evaluation of effects of plough cultivation and no-tillage on soil organic matter following conversion of a permanent pasture to arable cropping. The trial was on an Ohakea silt loam at Massey University and the crops grown were maize and oats. Soil sampling was to 10 cm. Total C at end of the trial was 34 t/ha with no-tillage and 28 t/ha with plough tillage.

Limitations: Major limitation is sampling depth of 10 cm. Since depth of cultivation was 20 cm, it is likely that C in the ploughed treatment was uniformly distributed over that depth, whereas C in the no-tillage treatment was stratified by depth with high concentrations near the soil surface. Therefore, shallow sampling for C likely led to a bias in favour of no-tillage.

Shepherd et al. 2001 - Effects of land use (pasture vs. arable cropping)

Shepherd et al. (2001) compared soil C under long-term (> 80 years) pasture with land that had been cropped for several years. In the Manawatu, a coarse textured Manawatu silt loam (well drained), a Kairanga silt loam (poorly drained), and a Moutoa humic clay (very poorly drained) were sampled under pasture and increasing duration of maize and/or barley cropping. In Taranaki, an Egmont soil (derived from andesitic ash) was sampled under pasture and 20 years of cropping (the paddock was double cropped each year with spring-sown barley and winter green-feed brassica). A Patumahoe clay (derived from weathered basalt and rhyolitic tephra) was sampled at Pukekohe under pasture and 40 years of vegetable production. All cropping sites were conventionally cultivated. Samples for C determination were taken from the 0-10 and 10-20 cm layers.

In all soils except Egmont, C declined under cropping. The rate of decline varied with the length of cultivation and soil type. In the Manawatu silt loam, rate of decline averaged 2.4 t C/ha during medium term (11 years) maize cropping, and slowed to 0.8 t/ha during 11 and 20 years of maize. In the finer-textured Kairanga soil, C declined at an average rate of 3.3 t/ha during short term (4 years of maize) cropping, and slowed to 0.9 t/ha during medium term (11 years of maize) and long-term (23 and 30 years of barley) cropping. The initial C loss in the fine textured Moutoa soil was negligible during short term (4 years of maize) cropping but increased to 1.1 t/ha between 4 and 11 years of maize. Under vegetable cropping at Pukekohe, C declined at an average rate of 1.2 t/ha over a 40 year period. Lowest rates of loss were observed on the allophanic Egmont soil (only 0.2 t/ha per year during 20 years of barley). On conversion back to (10-11 years) pasture, cropping soils gained C: 1.1 t/ha per year in the Manawatu, 0.9 t/ha per year in the Moutoa, and only 0.05 t/ha per year in the Kairanga soil.

APPENDIX 4: Datasets—Horticulture And Soil Carbon

4.1 APPLE ORCHARD SYSTEMS

DATABASE DESCRIPTION

Name of the data set:	Soil carbon management of two apple orchard systems (organic, integrated) in Hawke's Bay
Primary contact:	M Deurer, B Clothier (HortResearch Ltd, Palmerston North)
Data ownership / accessibility:	via HortResearch
Data storage:	Microsoft® Excel spreadsheets
Broad Land-use/Land form sampled:	Tree rows and alleys of two apple orchards
Number of sites:	Two. The sites are adjacent to each other. They have the same soil type and climate. Before their use as apple orchards, both sites were used for market gardening (a very similar initial soil organic carbon content can be assumed).
Geographical spread:	Havelock North, Hawke's Bay
Soil orders:	Recent
Depth of sampling: 0-30 cm:	Yes
Specific sampling depths:	0-10, 10-20, 20-30 cm depths
Bulk Density measurements:	Yes
Multiple sampling through time:	No
Pre-1990 soil sampling:	No
GIS reading of the site:	No
Soil description:	Yes
Are soil samples archived?	No
Total change in carbon (C) or rate of change in soil C recorded:	No
Land use history recorded:	Apple orchards for the last 10+ years preceding last sampling
Intensity of land use:	Recorded at the time of sampling. One orchard is under integrated, and the other under organic (BioGro) production.
Supplementary information on soils:	Information on various biophysical and chemical soil properties available (e.g., % C, % N, Olsen P, pH, water infiltration rates, aggregate stability, macropore topology, microbial activities, soil texture)
Management history:	Few details (especially for the organic) known (only generic information available)

Mitigation opportunities to minimise or improve soil C:

Evaluation of the potential of different generic soil carbon management for soil carbon sequestration: Within the tree rows, the two orchards differ in soil carbon management over the last 10+ years: On the integrated orchard the tree rows were regularly herbicided (no pasture or weeds), received no organic matter additions (apart from prunings) and were drip-

irrigated. On the organic orchard the tree rows had pasture, regularly received compost, and were not irrigated.

Conclusion: A soil carbon conservation management as practiced on the organic apple orchard can lead to significantly higher carbon contents in the topsoil (especially 0-10 cm) (Figure 4.1).

Associated publications:

Deurer M, Sivakumaran S, Ralle S, Vogeler I, McIvor I, Clothier B, Green S, Bachmann J 2008. A new method to quantify the impact of soil carbon management on biophysical soil properties: The example of two apple orchard systems in New Zealand. *J. of Env. Qual.* 37, 915-924.

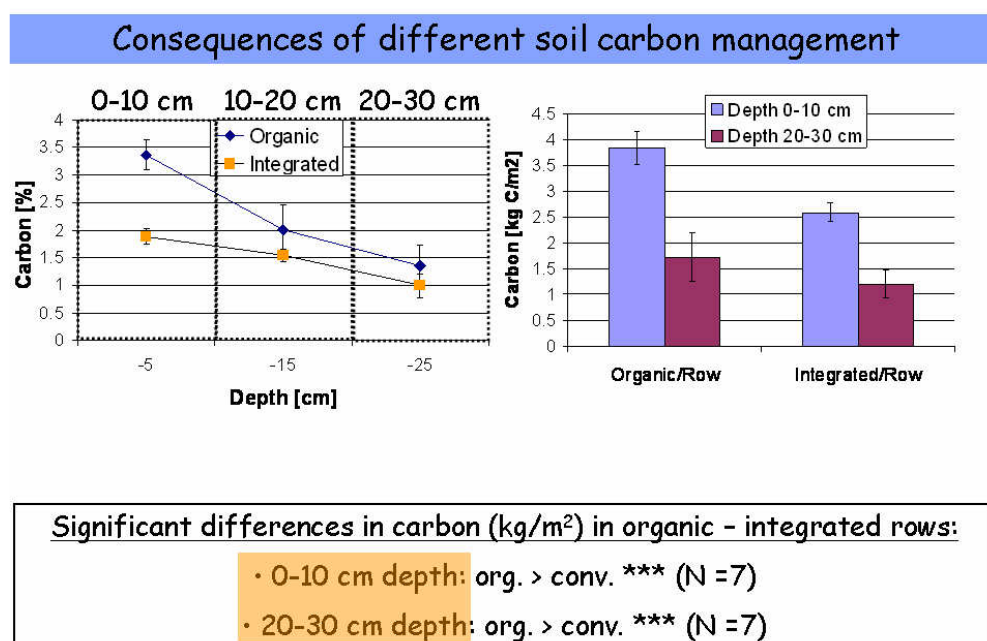


Figure 4.1. Soil carbon contents in the topsoil of an organic and an integrated apple orchard in Hawke's Bay. Both orchards are adjacent to each other and operated for 10+ years. They have the same soil type and climate and a very similar land use history.

4.2 VINEYARD SYSTEMS: MARLBOROUGH

DATABASE DESCRIPTION

Name of the data set:	Soil carbon status and management of vineyard systems in Marlborough
Primary contact:	M Greven (HortResearch, Blenheim)
Data ownership / accessibility:	HortResearch
Data storage:	Microsoft® Excel spreadsheets
Broad Land-use/Land form sampled:	Vineyard rows and inter-rows
Number of sites:	5 vineyards with 2-4 sites each, active database with new sites being added at the time of writing
Geographical spread:	Wairau Valley, with Awatere Valley to be added in next few months
Soil orders:	Recent
Depth of sampling:	0-15 cm
Specific sampling depths:	0-15 cm depth
Bulk Density measurements:	Yes
Multiple sampling through time:	No
Pre-1990 soil sampling:	No
GIS reading of the site:	No
Soil description:	No
Are soil samples archived?	No
Total change in C or rate of change in soil C recorded:	Yes when using headlands is used as a base line reference
Land use history recorded:	Yes
History for what period:	Varies per vineyard from 10-20 years
Intensity of land use:	Recorded
Supplementary information on soils:	Information on various biophysical and chemical soil properties available
Management history:	Little known (only generic information)

Mitigation:

We compared different soil carbon management (mulching v. no mulching, organic v. headland) on the same soil types, climate and general management. We found that mulching led to significantly higher carbon contents in the topsoils (Figure 4.2). Organic management with pasture in the rows led to soil carbon contents that were comparable with those of the headland (= situation of previous use as pasture) (Figure 4.2).

Comparing rows within the vineyard (same soil type, same climate) that were under vineyard use for different time lengths, we found a decrease of soil carbon in the vine row over 15 years (no longer times were available) (Figure 4.2).

Associated publications:

Deurer M, Greven M, Clothier B 2007. Carbon in horticultural soils – Digging for gold? In: “Carbon cost and climate change”, Proceedings of a workshop, Auckland, HortResearch, 7-8 May 2007.

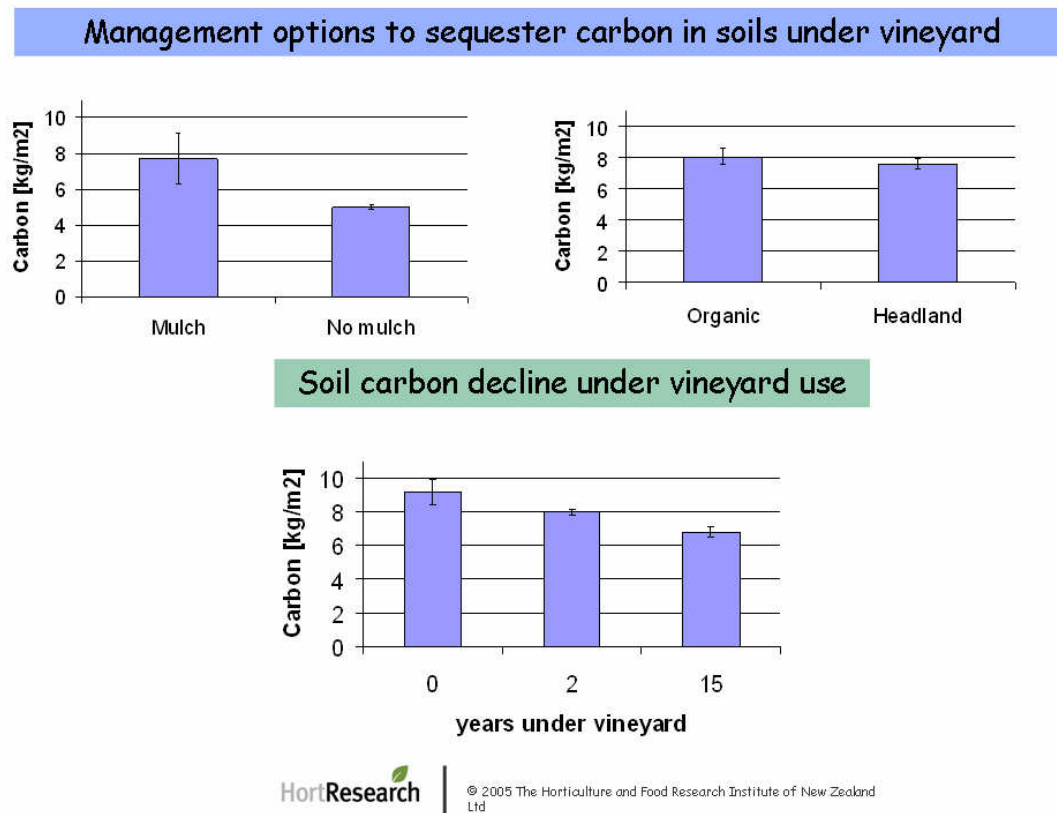


Figure 4.2. Conclusions from this preliminary study on soil carbon under vineyard use in Marlborough:

Top: Management options for carbon sequestration in soils under vineyard

Bottom: The decline of soil carbon over time under vineyard use.

4.3 VINEYARD SYSTEMS: HAWKE'S BAY

DATABASE DESCRIPTION

Name of the data set:	Soil carbon status of vineyards in Hawke's Bay
Primary contact:	T Mills, M Deurer (HortResearch, Palmerston North)
Data ownership / accessibility:	HortResearch
Data storage:	Microsoft® Excel spreadsheets
Broad Land-use/Land form sampled:	Wine grapes (Chardonnay, Cabernet)
Number of sites:	5
Geographical spread:	Hawke's Bay
Soil orders:	
Depth of sampling: 0-20 cm:	Yes
Specific sampling depths:	0-20 cm depth
Bulk Density measurements:	Yes
Multiple sampling through time:	No
Pre-1990 soil sampling:	No
GIS reading of the site:	No
Soil description:	Yes
Are soil samples archived?	No
Total change in C or rate of change in soil C recorded:	No
Land use history recorded:	Yes, some details available
History for what period:	Following information should be available: <ol style="list-style-type: none"> 1) When was it converted from pasture 2) Yield records 3) Irrigation or not 4) Fertiliser applications
Intensity of land use:	Yield recorded
Supplementary information on soils:	Mineral N at the time of sampling, C:N ratio, soil microbial dehydrogenase levels at time of sampling
Management history:	Little known (only generic information)
Associated publications:	None listed

4.4 KIWIFRUIT ORCHARDS: BAY OF PLENTY

DATABASE DESCRIPTION

Name of the data set:	Soil carbon status of kiwifruit orchards in the Bay of Plenty (4), Nelson (1) and Northland (1)
Primary contact:	T Mills, M Deurer (HortResearch, Palmerston North)
Data ownership / accessibility:	HortResearch
Data storage:	Microsoft® Excel spreadsheets
Broad Land-use/Land form sampled:	Pergola-grown ZESPRI™ GOLD kiwifruit ('Hort16A')
Number of sites:	6
Geographical spread:	Bay of Plenty, Nelson and Northland
Soil orders:	Recent (Nelson), Volcanically derived (Bay of Plenty-?) (Northland)
Depth of sampling: 0-20 cm:	Yes
Specific sampling depths:	0-20 cm depth
Bulk Density measurements:	Limited (Te Puke sandy loam)
Multiple sampling through time:	No
Pre-1990 soil sampling:	No
GIS reading of the site:	No
Soil description:	Yes
Are soil samples archived?	No
Total change in C or rate of change in soil C recorded:	No
Land use history recorded:	Yes (good option for some generic information to be pulled together for kiwifruit management)
If yes, for what period?	Probably not too difficult to obtain: <ol style="list-style-type: none"> 1) When was it converted from pasture 2) Yield records 3) Irrigation or not 4) Fertiliser applications
Intensity of land use:	Recorded
Supplementary information on soils:	Have information on mineral N at the time of sampling, C:N ratio, soil microbial dehydrogenase levels at time of sampling,
Management history:	Little known (only generic information)