Ministry for Primary Industries
Manatū Ahu Matua

Review of the impacts of climate change on soil processes and the consequences for ecosystem services - Appendices

MPI Technical Paper No: 2018/41

Prepared for the Ministry for Primary Industries
by Soil and Land Use Alliance (SLUA)

ISBN No: 978-1-77665-921-0 (online)
ISSN No:

June 2013

New Zealand Government

Disclaimer

Requests for further copies should be directed to:

Publications Logistics Officer
Ministry for Primary Industries
PO Box 2526
WELLINGTON 6140

Email: brand@mpi.govt.nz
Telephone: 0800 00 83 33
Facsimile: 04-894 0300

This publication is also available on the Ministry for Primary Industries website at http://www.mpi.govt.nz/news-resources/publications.aspx

Enquiries to:
Bryan Stevenson: StevensonB@landcareresearch.co.nz
Loretta Garrett: Loretta.garrett@scionresearch.com
Tony van der Weerden: tony.vanderweerden@agresearch.co.nz

Report reference:
SLUA (2013). Review of the impacts of climate change on soil processes and the consequences for ecosystem services – Appendices. Report prepared for MPI, June 2013.

Authors:
B Stevenson[2], L Garrett[3], T van der Weerden[1], M Beare[4], P Beets[3], N Bell[1], S Bowatte[1], B Clothier[4], D Curtin[4], C de Klein[1], M Davis[3], I Dickie[2], M Dodd[1], E Dominati[1], R Gentile[4], C Hedley[2], M Kirschbaum[2], B Mullan[5], M O'Callaghan[1], K Orwin[2], R Parfitt[2], M Shepherd[1], S Smaill[3], S Thomas[4], S Wakelin[1].

[1]AgResearch, [2]Landcare Research, [3]Scion, [4]Plant & Food, [5]NIWA

Contents

Appendix 1 – Technical review

1 Introduction

This technical review supports the discussions in the main body of the report. It is a source of more detailed discussion (and references), and was distilled down for the main body of the report. The sections contained within the technical review covered the three topics within the SLMACC (Sustainable Land Management and Climate Change) Climate Change Effects on Soils theme (soil carbon, soil services and soil nitrogen), and represent a compilation of the findings of these three groups. The evidence for the climate change projections for New Zealand is also presented in this technical review. The individual reviews were undertaken using available literature and combined with current best knowledge from national experts. The national experts were brought together at a workshop in February 2013 to investigate and debate the evidence in order to be more certain of the outcomes of climate change on soils.

When interpreting the effects of climate change on soil carbon, soil services and soil nitrogen, it is important to factor in the experimental approach (type and design). Experiments to inform climate change effects have been undertaken at a range of scales from small pot experiments to large plot scale Free Air CO_2 Enrichment (FACE) experimental installations and include one or more variables in the experimental design.

FACE experiments (the nearest to a fully integrated system) are relatively few globally; even then, responses are likely to be site specific to the experimental conditions. Experiments also tend to look at the effects of large increases in CO_2 whereas, in reality, systems are already responding to gradual elevations in CO_2 rather than having to adapt rapidly to a large change in elevated CO_2 (eCO_2). There is a paucity of studies that have looked at the combined effects of increases in eCO_2 and temperature and changes in precipitation.

A large proportion of the literature focused on natural ecosystems and/or where external inputs such as nitrogen (N) via ‘deposition’ are relatively small. Whilst this will be relevant for low input systems, results require some extrapolation to more intensive systems where fertiliser inputs tend to be larger. A further complication in interpreting the larger scale effects from experiments is that they may indicate an effect at the plant or micro-plot scale but the larger scale response (paddock/farm) might be modified by other system responses.

With these experimental limitations at play, we have reviewed the available literature and incorporated the views of national experts to provide a discussion on the effects of climate change on soil processes and the consequences on soil ecosystem services within New Zealand.

2 Climate change projections for New Zealand

Brett Mullan[1]

NIWA[1]

2.1 KEY MESSAGES

Future changes in the climate are not known with certainty. Projected changes in climate depend on the assumptions made about the impact of greenhouse gases. It is important to examine different scenarios for their likely impacts on soil processes. Table 2.1 summarises the range of projected changes by mid-century and end-of-century.

Table 2.1: Predicted ranges for key climate variables based on diverging scenarios of carbon dioxide increase (1). The range in values provided a guide for magnitude of climate change effects we assess in the report. Changes are relative to 1980-1999 levels.

Variable	Season	Region of NZ	Range predicted for year 2049	Range predicted for year 2099	Level of confidence in predicted values
Carbon dioxide * (ppm)	All	All	480 to 530 ppm	450 to 850 ppm	Moderate to high
Temperature (°C)	All	All	0.7 to 0.9	1.1 to 2.6	High
Change in rainfall (%)	Summer & Autumn	South & west S.Is. Rest of NZ.	Zero to +5% Up to ±5%	Zero to +5% Up to ±5%, & >+5% in eastern Nth Island	Moderate
	Winter & Spring	North & east N.Is., Marlborough, Canterbury Plains West N.Is., south & west S.Is.	Zero to -10% Zero to +10%	-5 to -20% Zero to +30%	High
Hot Days	Summer half-year	All lowland areas	Up to 100% increase	Up to 300% increase	High
Frosts	Winter half-year	Central N.Is. & S.Is.	Up to 50% reduction	Up to 50-90% reduction	High
Heavy rainfall	All	Especially in west of both Islands & south of S.Is.	Extremes occur up to 50% as often	Extremes occur up to 100% (i.e. 2 times) as often	High
Drought	Summer half-year	Mainly eastern areas Eastern S. Is. & all of N. Is.	Up to 5-10% more of year	 At least 10% more of year	Moderate for type of change; low for magnitude
Strong winds	Winter, Spring	All	Increase of few %	Increase up to ~10% in frequency	Moderate for type of change; low for magnitude
		N. Is.	Little change	Little change	
	Summer/ Autumn	S. Is.	Decrease of few %	Decrease of few per cent	

(1) From the "high carbon" and the "rapidly decarbonising" world scenarios
* Current level of carbon dioxide is 395 ppm

In terms of confidence in the projections, there is high confidence in the temperature increase, the changes in temperature extremes (more hot days, fewer frosts) and the increase in heavy rainfall. There is high confidence in the pattern of winter/spring rainfall change, and but only moderate confidence in the summer/autumn pattern. Other changes in extremes (drought, strong winds) have moderate confidence in the direction of change but low confidence in the magnitude of change.

2.2 INTRODUCTION

There is a wide range of scenarios describing how the world may develop in terms of social, economic and technological change over the 21st century (Meinshausen et al., 2009), and a large number of climate model projections of future climate change (Meehl et al., 2007). NIWA has produced various analyses and summaries of future projections for New Zealand (MfE, 2008; Mullan et al., 2011; Clark et al., 2011). These ‘what if’ scenarios and projections are used because we cannot predict what socio-economic choices the world will make, and thus what future greenhouse gas emissions will be or the subsequent details of climate change. For the purpose of this study, two scenarios were selected that represent likely extremes of future emissions and global climate changes: (1) a high carbon world, and (2) a rapidly decarbonising world. These scenarios were developed for an international conference organised by the New Zealand Climate Change Centre in 2009, and are described in Reisinger et al. (2010). By the end of the century, global average temperatures would be about 4°C above pre-industrial under the high carbon world scenario, whereas the temperature increase would be limited to about 2°C under the rapidly decarbonising world scenario.

A brief description of the scenarios is provided below, with full technical details in Reisinger et al. (2010). The New Zealand climate change patterns are derived by statistically downscaling 12 global climate models (GCMs), the results of which were discussed in the IPCC Fourth Assessment Report (IPCC, 2007). These 12 models were selected because their 20th century simulations validated well against current climate in the New Zealand and Southwest Pacific region (MfE, 2008; Mullan and Dean, 2009). This present study emphasises the likely range between the extreme scenarios of the high carbon and rapidly decarbonising worlds, with a particular focus on the 12-model average. Ensemble averages are commonly used in seasonal prediction as being more reliable than individual models (Johnson and Bowler, 2009), and model averages tend to validate better on historical climate than any individual model (Mullan and Dean, 2009). In the Reisinger et al. (2010) study, downscaled climate changes were calculated on NIWA’s Virtual Climate Station (VCS) 5-km grid over New Zealand (Tait et al., 2006). The climate scenarios were expressed as changes between the 20-year current climate period 1980-1999 and the future 20-year periods of 2030-2049 and 2080-2099. This 20-year averaging removes much, but not all, of the natural variability as represented in the models. The same time periods and terminology are retained here for this report. Therefore, it is important to note that a reference to a “2049 change”, for example, means the change between 1980-1999 and 2030-2049.

The following section first describes projected changes in 12-model averages of temperature and precipitation, then discusses likely changes in extremes. Potentially, the biggest impact of future climate change will occur with extreme events such as heavy rainfall and flooding, droughts, strong winds and high temperatures. However, extremes were not analysed explicitly for the high carbon and rapidly decarbonising scenarios, so this discussion is necessarily more qualitative.

2.3 CHANGES IN CARBON DIOXIDE CONCENTRATIONS, TEMPERATURE AND PRECIPITATION

Carbon dioxide concentrations under the two scenarios

The high carbon world scenario is based on the SRES A2 emission scenario (IPCC, 2000), with associated climate changes over the 21st century that are about eight times larger and faster than those observed already over the 20th century (in terms of global average temperature change). This scenario represents a highly fractured world with no concerted efforts to reduce greenhouse gas emissions, heterogeneous socio-economic and technological development in different parts of the world, large rates of climate change, warming of almost 4°C above pre-industrial levels, and severe impacts resulting in many regions of the world (Reisinger et al., 2010).

The rapidly decarbonising world scenario is based on the SRES B1 emissions up until mid-century, followed by concerted global action to reduce greenhouse gas emissions and to limit greenhouse gas and aerosol concentrations to about 450 ppm CO_2-equivalent. This results in a stabilisation of global temperatures by the end of the century at about 2°C above pre-industrial levels (but note that sea-level rise does not stabilise over this time-frame).

Figure 2.1 shows the evolution during the 21st century of atmospheric CO_2 concentration under the Representative Concentration Pathways (RCPs) developed for the IPCC Fifth Assessment (Meinshausen et al., 2011). The CO_2concentrations used in this report are indicated by the diamonds in Fig. 2.1; the mid and end of century concentrations for the high carbon and rapidly decarbonising scenarios fit reasonably well into the overall range of the newer RCP scenarios (NIWA has yet to assess GCM results derived from these Fifth Assessment scenarios). The divergence in CO_2 concentration between the scenarios is pronounced by century end, but is only modest at the mid-century point.

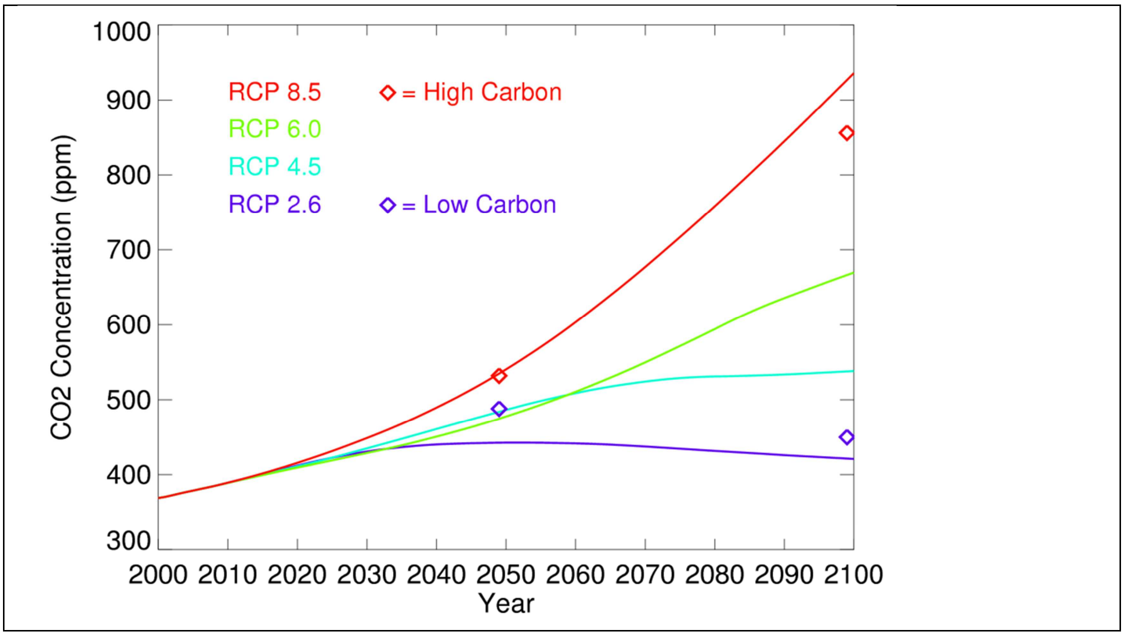

Fig. 2.1: Trajectories of atmospheric CO_2 concentration from 2000 to 2100 under the four RCP scenarios, compared to a modification of the SRES scenarios as used in this report. Diamonds mark the mid and end of century concentrations for the high carbon (red) and rapidly decarbonising (or 'low carbon'; blue) SRES scenarios of Reisinger et al. (2010).

Model-average changes in temperature and precipitation under the two scenarios

Maps of temperature changes projected for New Zealand under the two emissions scenarios are presented in Figure 2.2. The spatial gradients in temperature change are weak, which is partly a consequence of smoothing that is implicit in the statistical downscaling algorithm. The 12-model average displays slightly greater warming in the north of New Zealand than in the south. Only the annual changes are presented here, since there is little variation seasonally in the projected temperature changes. For example, in the North Island, the annual temperature increase by the end of the century under the high carbon world is more than 2.5°C, but the seasonal differential is only about 0.3°C (with summers warming slightly more than winters).

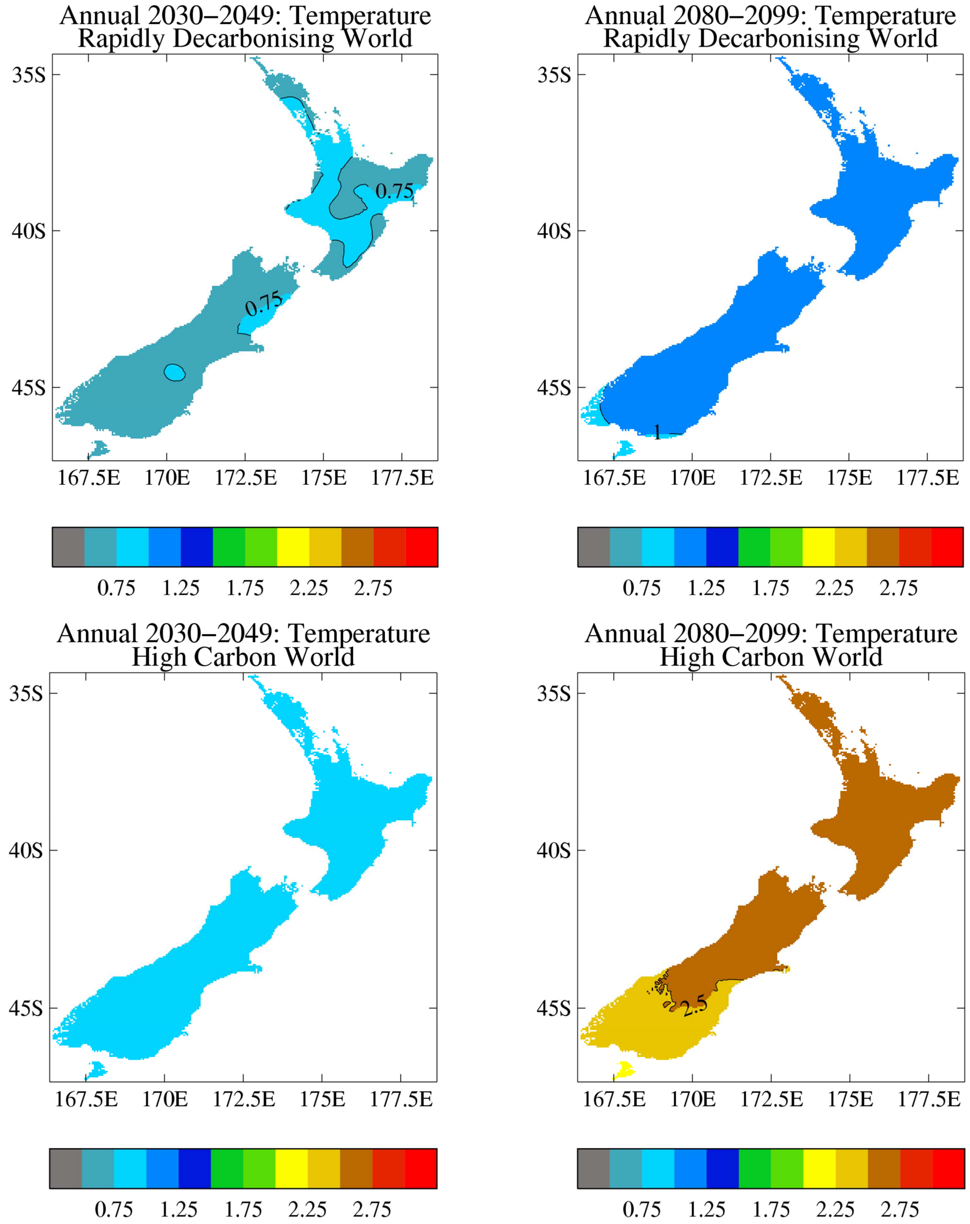

Figure 2.2: Projected New Zealand annual temperature change (in °C) from 1980-1999 to 2030-2049 and to 2080-2099 for the rapidly decarbonising world (upper panels), and for the high carbon world (lower panels), as averaged over the downscaled patterns from 12 global models (from Reisinger et al., 2010).

Figures 2.3 and 2.4 show the 12-model average patterns of projected precipitation change at 2030-2049 and 2080-2099 for the two scenarios. Because of substantial seasonal differences in precipitation changes, maps are shown for both summer and winter. In general, autumn changes are similar to those in summer, and spring similar to winter.

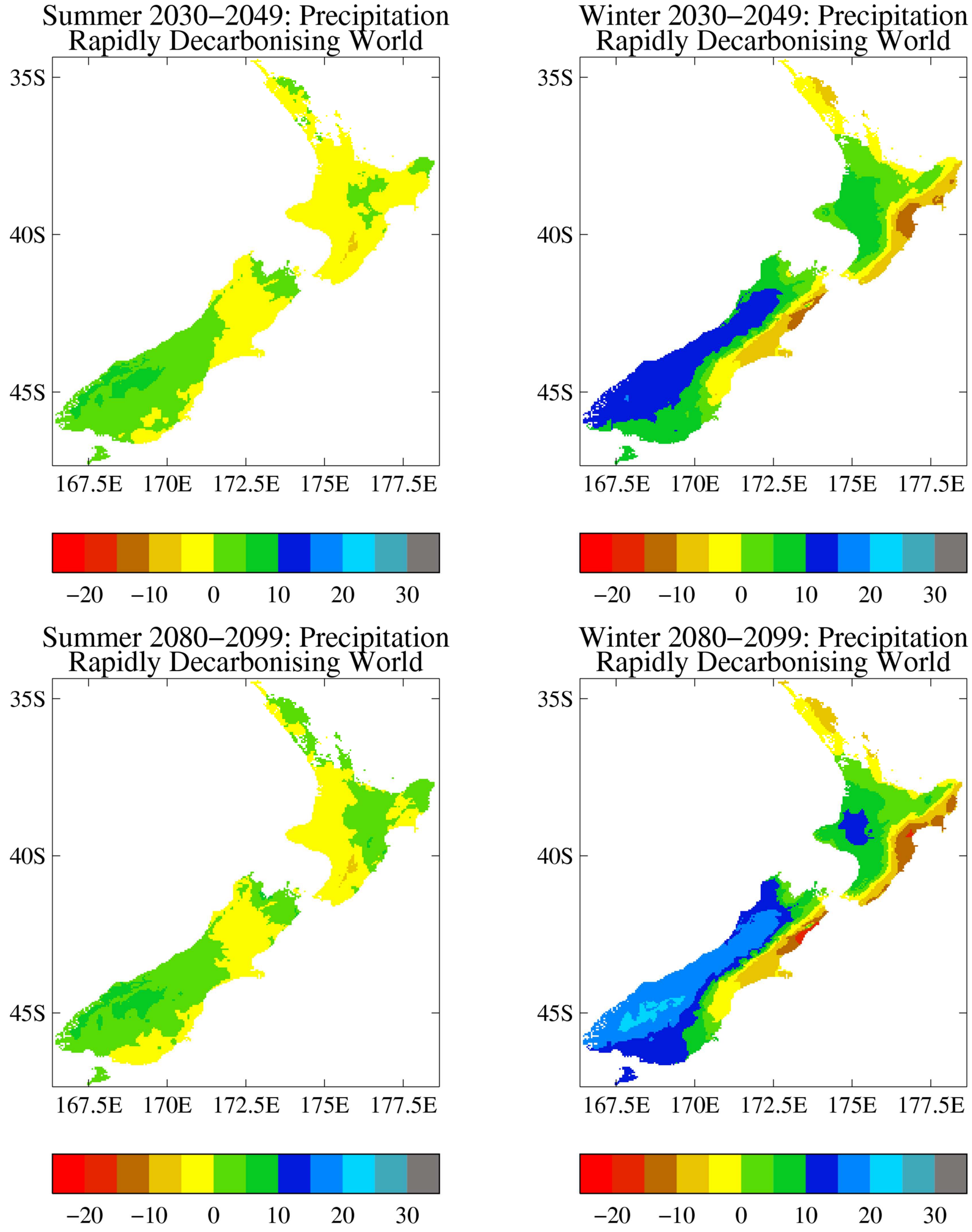

Figure 2.3: Projected New Zealand seasonal precipitation change (in %) from 1980-1999 to 2030-2049 (upper panels), and to 2080-2099 (lower panels), for the rapidly decarbonising world as averaged over the downscaled patterns from 12 global models. Summer and winter seasons are shown.

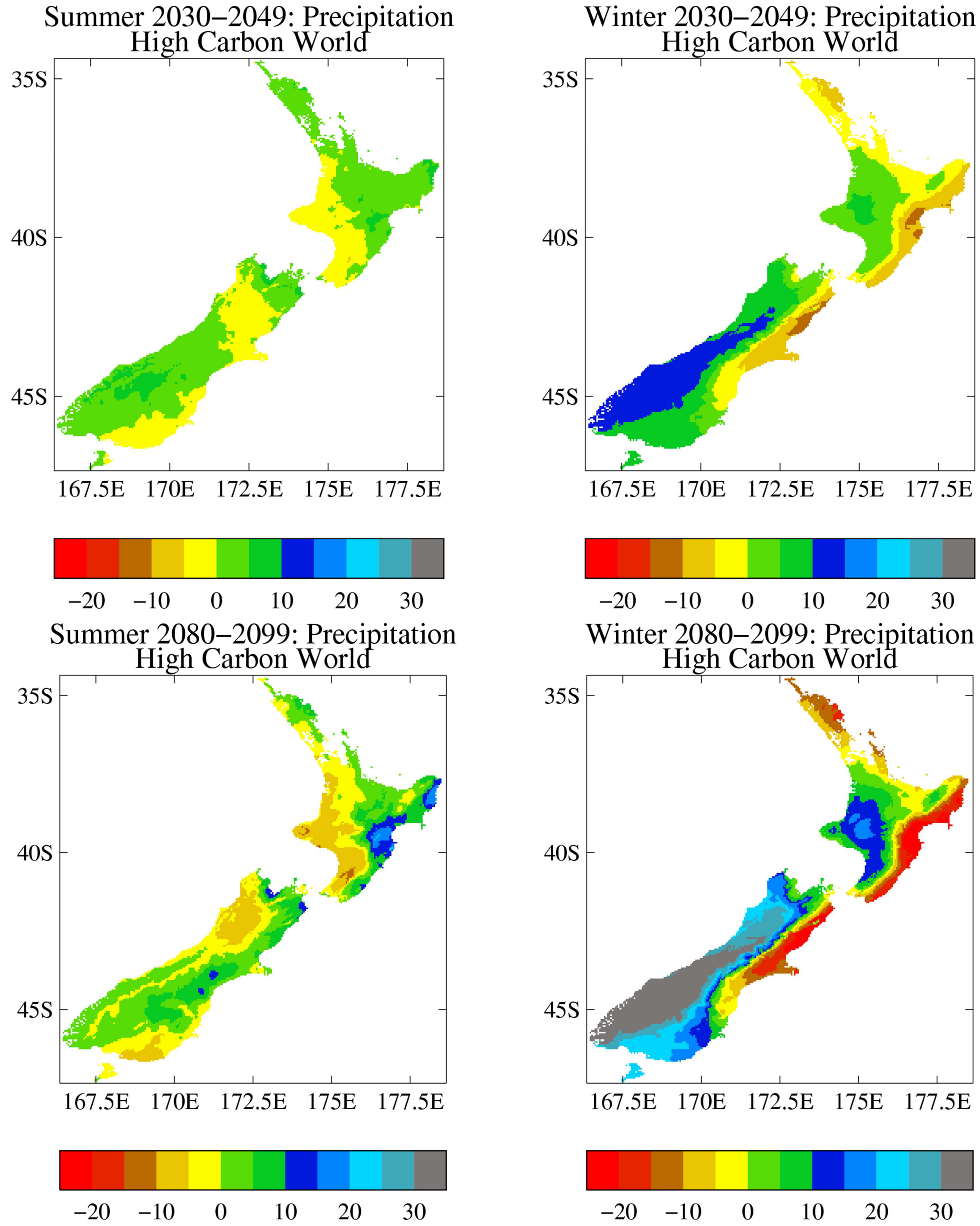

Figure 2.4: As for Figure 2.3, but for the high carbon scenario.

Winter changes have a consistent pattern of increases in the west of both islands, and decreases in the east of both islands and in the north of the North Island. The amplitude of this pattern increases with time and with the magnitude of global warming. This strong west-east gradient in precipitation change is driven by the increase in southern hemisphere westerly winds, which is a very consistent feature across virtually all global climate models.

Summer precipitation changes are generally smaller than those of winter, and lie within ±5% over almost the whole country for both periods of the rapidly decarbonising scenario, and for the first 50 years of the high carbon world. The pattern of summer rainfall change in the North Island, however, is noticeably different from winter: a decrease is projected for the southwest and an increase in the east, in the 12-model average. This pattern becomes more marked under the high carbon world by the end of the century (Fig. 2.4, lower left panel) with up to 10% less summer rainfall in Taranaki, Wanganui and Manawatu, and increases of 10% or more in parts of Hawke's Bay and Gisborne by 2080-2099. These summer changes in the North Island are the opposite of those in winter, but smaller, so the winter pattern will dominate the annual average.

Inter-model variations and natural decadal variability

Figures 2.2-4 illustrate future changes as averaged over 12 models. The potential range of projected changes is obviously larger if individual models are considered. Figure 2.5 gives an example for projected rainfall changes by the end of the century under the low and high carbon scenarios. The summer distributions are quite wide, but the averages lie close to zero. Conversely, the winter distributions are narrower, significantly positive, and greater for the high carbon scenario.

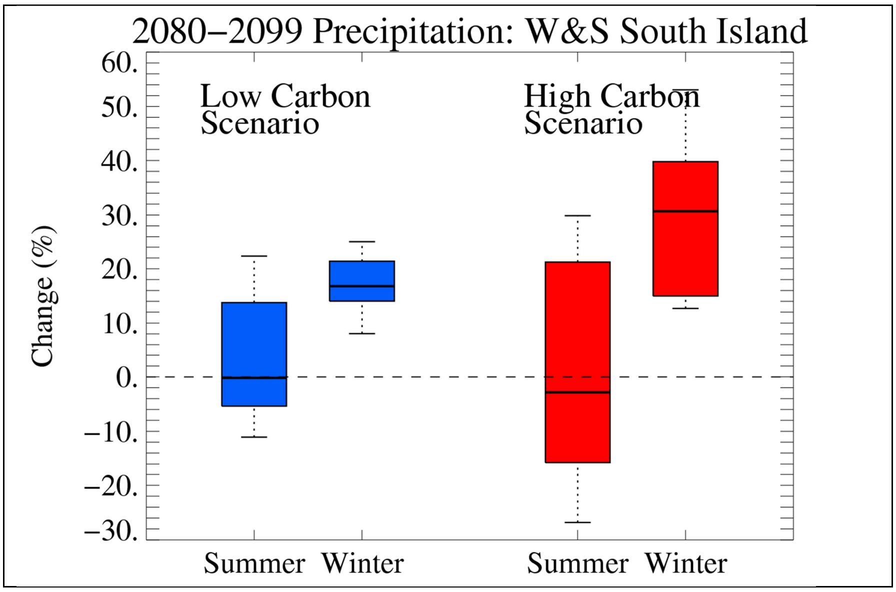

Figure 2.5: Box and whisker plots showing the distribution of precipitation changes across 12 GCMs, as projected for the west and south of the South Island (blue region in Fig. 2.5): rapidly decarbonising (blue plots) and high carbon (red plots) scenarios.

Figure 2.6 helps to place the projections of rainfall change in the context of natural decadal variability. A climatic phenomenon known as the Interdecadal Pacific Oscillation (IPO) has noticeable effects on long-term climate variations in New Zealand and other places in the Pacific. The positive phase of the IPO is associated with more frequent El Niño events in the Pacific and a weather regime of more frequent westerly winds over New Zealand – leading to increased rain in the west and reduced rainfall in the east (Fig. 2.6). The negative IPO phase is associated with more frequent La Niña events, weaker westerlies over New Zealand, and the opposite rainfall pattern.

The positive phase IPO has the same pattern as the climate change signal, i.e., wetter in the west and drier in the east. The amplitude of the IPO-rainfall pattern is similar to that at mid-century under either CO_2 scenario. Adding the IPO signal to the climate change signal could therefore double or negate the intensity of the rainfall anomaly pattern at mid-century. Although the IPO phase appears fairly persistent over a decade or two, the mechanism is not fully understood and we cannot predict the phase at mid-century.

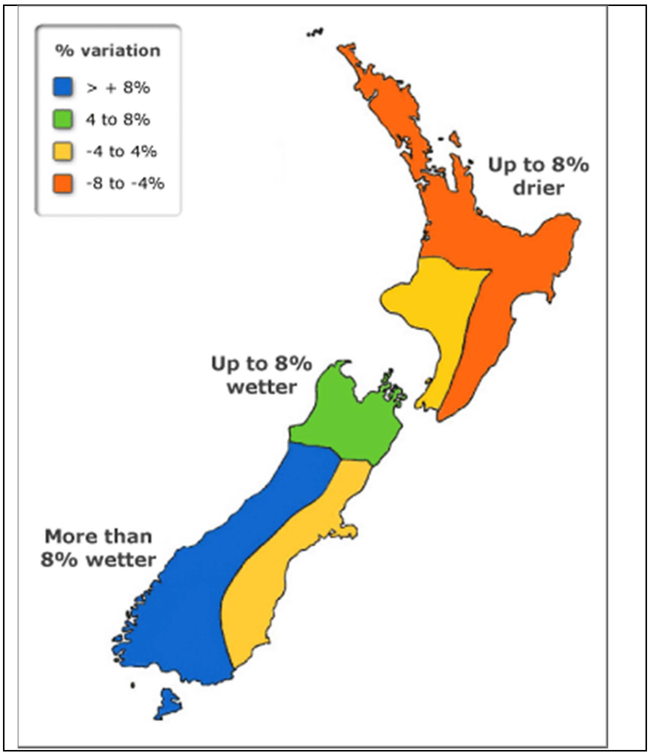

Figure 2.6: Precipitation change (%) between the negative and positive phase of the Interdecadal Pacific Oscillation. Changes are shown for the six seasonal outlook regions used by NIWA, and represent the average changes for 21-year periods before and after 1977.

2.4 CHANGES IN EXTREMES

Temperature extremes

Fewer frosts and increases in high maximum temperatures are probably the most robust findings from analyses of climate change simulations (e.g., Kharin et al., 2007). Indicative results for New Zealand can be found in the referenced 2008 MfE publication. Many parts of lowland New Zealand are likely to become virtually frost-free during the 21st century. New Zealand does not experience the extreme high temperatures found in Australia, for example, so most NIWA analyses have focussed on so-called ‘hot days’, defined as exceeding 25°C. Under the high carbon scenario, many locations that now have fewer than 30 ‘hot days’ per year could experience a 3 or 4-fold increase by the end of the century.

Precipitation extremes

Increases in the frequency and magnitude of heavy rainfall are also a widespread finding of climate models. The potential low level moisture content of the atmosphere rises at about 7-8% for each 1°C increase (MfE, 2008; Allan & Soden, 2008). Current guidance for New Zealand (MfE, 2008) suggests that under a local warming of about 2°C, extreme rainfall could occur approximately twice as often (e.g.,a 24-hour extreme that currently has a 100-year return period could recur every 50 years) Figure 2.7 maps the current 24-hour rainfall accumulations associated with a 100-year return period. With the exception of central Otago, 100-year daily rainfall extremes are at least 100mm, and increase to more than 500mm in Westland, Fiordland, and isolated pockets in the North Island. At Christchurch Airport, for example, the current 24-hour 100-year extreme is about 136 m. Under 2°C warming (the high carbon scenario by century end), this amount of rainfall could occur every 50 years and the 100-year extreme increase to 157 mm (MfE, 2008). Increasing flood peaks resulting from increased precipitation extremes will interact with rising sea levels and could create particular challenges for some coastal settlements or low-lying areas.

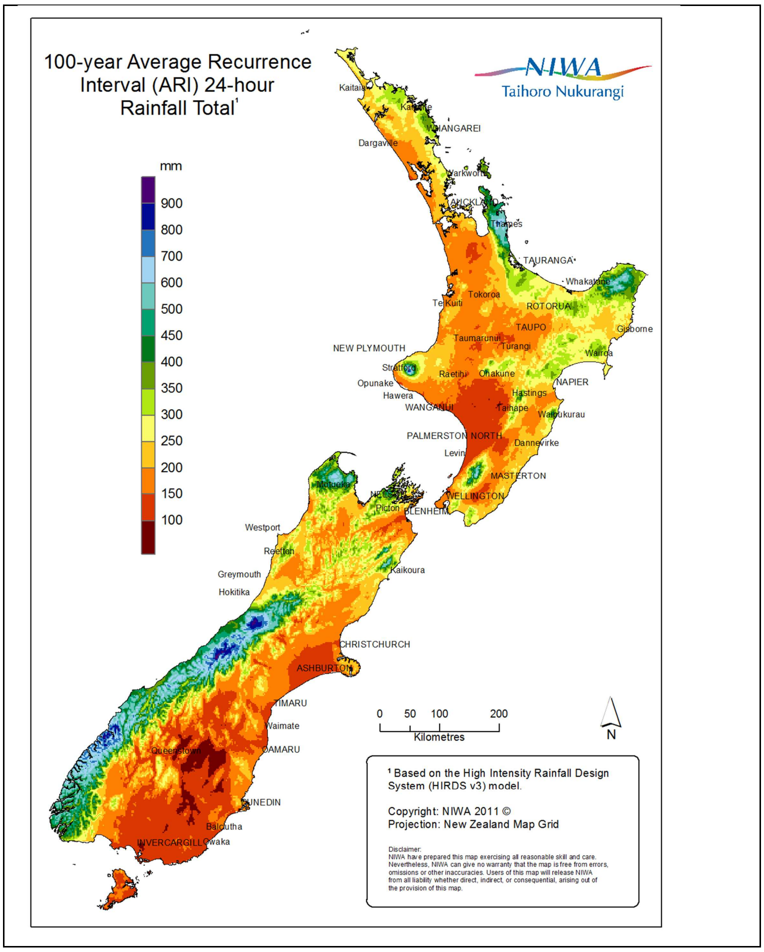

Figure 2.7: Calculated 24-hour extreme rainfall depths with a return period of 100 years under the current climate. (From Basher et al., 2012.)

Drought

Drought magnitude and frequency are expected to increase in a warmer climate as evapotranspiration increases, unless this increase is compensated by a simultaneous increase in precipitation. Mullan et al. (2005) concluded from a study of two IPCC Third Assessment models that the current 1-in-20 year potential evapotranspiration (PET) deficit (defined as a 'drought') could occur at least twice as often in eastern parts of New Zealand (parts of Northland, Bay of Plenty, Wairarapa, Marlborough, Canterbury and Otago) under a warming of about 2°C. Figure 2.8 is taken from a more recent NIWA study of drought trends under climate change, and finds a similar pattern (although with large uncertainty depending on the model used and the methodologies of downscaling and evapotranspiration calculation).

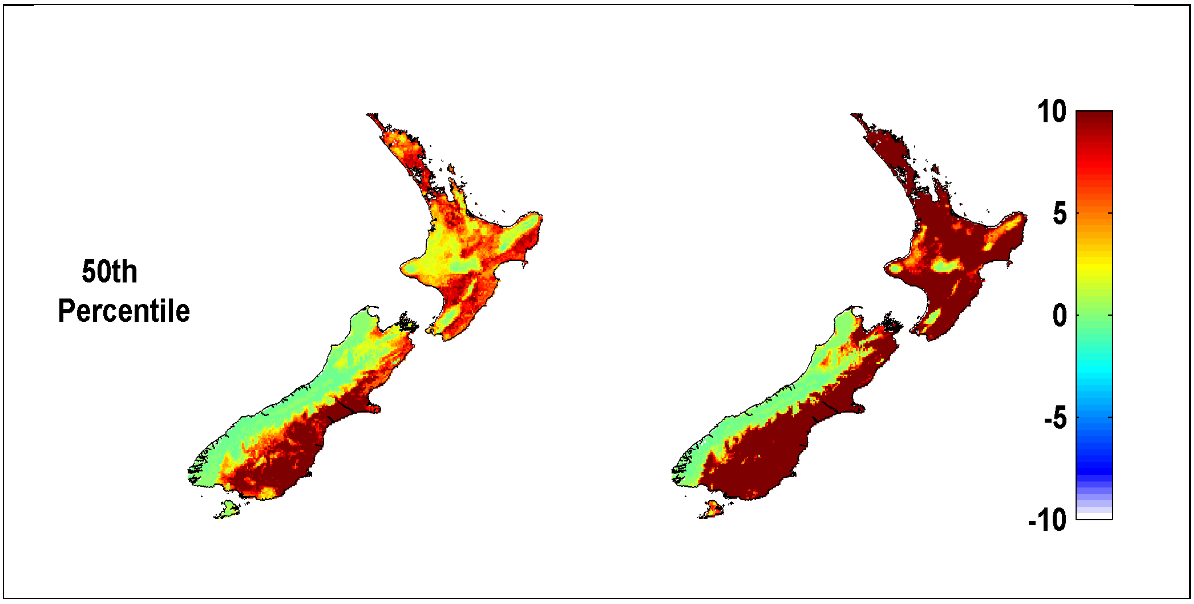

Figure: 2.8: Projected increase in % of year in drought for the SRES A2 emission scenario (from Clark et al., 2011). Changes are shown for the 50th percentile across 19 climate models, for the period 2030-2050 (left) and 2070-2090 (right), relative to 1980-1999 levels.

Extreme winds

Some international studies have suggested an increase in the frequency of strong winds under global warming (e.g., Rockel and Woth, 2007), but the change is very dependent on geographic location. The MfE (2008) climate change guidance manual suggested that up to a 10% increase in the strongest winds (top 1-percentile) is possible by 2090 under the stronger SRES emission scenarios.

A recent New Zealand study (Mullan et al., 2011) presents a more complex picture: it suggests extreme wind speeds at the large-scale (i.e., excluding thunderstorms and frontal systems) might increase only a few percent by the end of the century under the middle-of-the-road SRES A1B emission scenario. However, the seasonal distribution was likely to change, with an increase in the frequency (but not necessarily extreme magnitude) of extreme winds in almost all regions in winter, but a decrease in summer in the Wellington region and the South Island.

Figure 2.9 (left-hand panel) gives an example from the Mullan et al. (2011) study, showing a decrease in pressure (i.e., deeper and more intense lows) over and south of the South Island in the winter season. The result was derived by tracking cyclone centres from daily pressure data, available from five GCMs for 20th and 21st century periods. Figure 2.9 (right-hand panel) shows a separate result from same study pertaining to extreme winds at the thunderstorm scale. The figure shows a map of changes in extreme daily values of the modified K Index (calculated from the vertical profile in temperature and humidity), which weather forecasters find useful for predicting the likelihood of severe convection and thunderstorm activity. The 99th percentile in this convective index increases everywhere in the New Zealand domain (as calculated from NIWA's Regional Climate Model (RCM) under a SRES A2 scenario), suggesting a future large-scale environment more conducive to thunderstorm activity. However, the RCM does not run at a high enough resolution to simulate such extreme events directly, so no quantitative projections of extreme winds at the small scale can be made at this time.

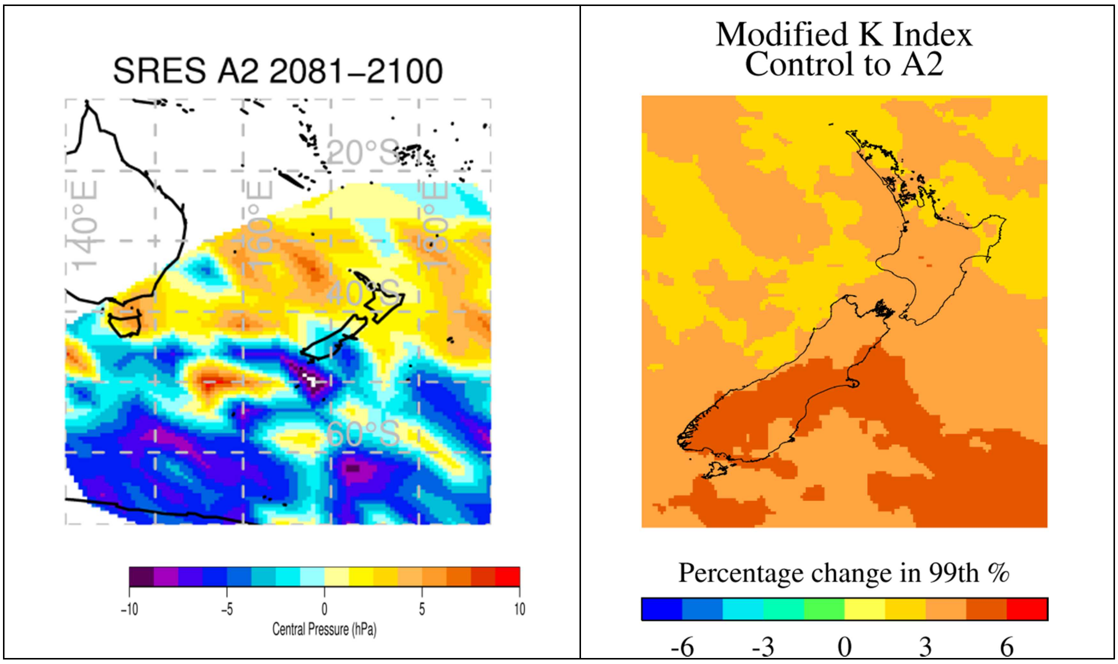

Figure 2.9: Change (hPa) in mean winter cyclone central pressure, averaged over 5 GCMs (left), and percentage change in the 99th percentile of the daily modified K index from NIWA's regional climate model (right), comparing 20-year periods at the end of the 20th and 21st centuries (from Mullan et al., 2011).

Fire risk

A study by Pearce et al. (2005), adopting the moisture scenarios of Mullan et al. (2005), found increases in a range of fire indices, especially in eastern parts of New Zealand, that were more marked under the higher warming scenarios.

Pearce et al. (2005) used two global climate models from the IPCC 3rd Assessment with contrasting spatial patterns of climate change were adopted, and used to modify observed fire danger records at 52 Fire Service sites under low, mid-range and high climate change scenarios. For both models under mid-range and high scenarios, the total number of days of very high or extreme forest fire danger increased by more than 50% (to 20 days) at several eastern sites in the North and South Islands. The model scenarios also suggested that fire season length could be extended in many parts of the country.

2.5 REFERENCES

Allan, R.P., Soden, B.J. (2008). Atmospheric warming and the amplification of precipitation extremes. *Science* 321, 1481-1484.

Basher, L., Elliott, S., Hughes, A., Tait, A., Page, M., Rosser, B., McIvor, I., Douglas, G., Jones, H. (2012). *Impacts of climate change on erosion and erosion control methods – A critical review.* MPI Technical Paper No. 2012/45.

Clark, A., Mullan, B., Porteous, A. (2011). *Scenarios of regional drought under climate change.* Client Report WLG2010-32 for Ministry of Agriculture and Forestry, 135p. June 2011.

IPCC. (2000). *Special Report on Emissions Scenarios*. Nakicenovic, N., Swart, R. (Eds). Cambridge University Press, Cambridge, United Kingdom, 104 p.

IPCC. (2007). *Climate Change 2007: The Physical Science Basis.* Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Solomon S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L., (Eds). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 996 p.

Johnson, C., Bowler, N. (2009). On the reliability and calibration of ensemble forecasts. *Monthly Weather Review 137*, 1717-1720.

Kharin, V.V., Zwiers, F.W., Zhang, X., Hegerl, G.C. (2007). Changes in temperature and precipitation extremes in the IPCC ensemble of global coupled model simulations. *Journal of Climate 20*, 1419-1444.

Meehl, G.A., Stocker, .TF., Collins, W.D., Friedlingstein, P., Gaye, A.T., Gregory, J.M., Kitoh, A., Knutti, R., Murphy, J.M., Noda, A., Raper, S.C.B., Watterson, I.G., Weaver, A.J., Zhao, Z.-C. (2007). Global Climate Projections. In: *Climate Change 2007: The Physical Science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (eds.). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Meinshausen, M., Meinshausen, N., Hare, W., Raper S.C.B., Frieler K., Knutti R., Frame D.J., Allen M.R. (2009). Greenhouse-gas emission targets for limiting global warming to 2°C. *Nature, 458*, 1158-1162.

Meinshausen, M., Smith, S.J., Calvin, K., Daniel, J.S., Kainuma, M.L.T., Lamarque, J.F., Matsumoto, K., Montzka, S.A., Raper, S.C.B., Riahi, K., Thomson, A., Velders, G.J.M., van Vuuren, D.P.P. (2011). The RCP greenhouse gas concentrations and their extensions from 1765 to 2300. *Climatic Change*, 109: 213-241.

Ministry for the Environment. (2008). *Climate Change Effects and Impacts Assessment. A Guidance Manual for Local Government in New Zealand.* 2nd Edition. Prepared by Mullan, B., Wratt, D., Dean, S., Hollis, M. (NIWA); Allan, S., Williams, T. (MWH NZ Ltd), Kenny, G. (Earthwise Consulting Ltd). NIWA Client Report WLG2007/62, February 2008, 156p.

Mullan, B., Carey-Smith, T., Griffiths, G., Sood, A. (2011). *Scenarios of storminess and regional wind extremes under climate change.* Client Report WLG2010-31 for Ministry of Agriculture and Forestry, 80p., March 2011.

Mullan, B., Dean, S. (2009) *AR4 climate model validation and scenarios for New Zealand.* Presented at 9th International Conference on Southern Hemisphere Meteorology and Oceanography, Melbourne, 9-13 February 2009.

Mullan, A.B., Porteous, A., Wratt, D., Hollis, M. (2005). *Changes in drought risk with climate change.* NIWA report to Ministry for the Environment, May 2005.

Pearce, H.G., Kerr, J., Clark, A., Mullan, B., Ackerley, D., Carey-Smith, T., Yang, E. (2011). *Improved estimates of the effect of climate change on NZ fire danger.* Scion, Rural Fire Research Group, Christchurch, in conjunction with NIWA, Wellington. Scion Client Report No. 18087. 84 p.

Pearce, H.G., Mullan, A.B., Salinger, M.J., Opperman, T.W., Woods, D., Moore, J.R. (2005). *Impact of climate change on long-term fire danger.* Report by Forest Research and NIWA to NZ Fire Service Commission, May 2005.

Reisinger, A., Mullan, A.B., Manning, M., Wratt, D.W., Nottage, R.A.C. (2010). *Global and local climate change scenarios to support adaptation in New Zealand. In:* Climate change adaptation in New Zealand: Future scenarios and some sectoral perspectives. Nottage, R.A.C., Wratt, D.S., Bornman, J.F., Jones, K. (eds). New Zealand Climate Change Centre, Wellington, pp 26-43.

Rockel, B., Woth, K. (2007). Extremes of near-surface wind speed over Europe and their future changes as estimated from an ensemble of RCM simulations. *Climatic Change 81*, 267-280.

Salinger, M.J., Renwick, J.A., Mullan, A.B. (2001). Interdecadal Pacific Oscillation and South Pacific climate. *International Journal of Climatol*ogy *21*, 1705-1721.

Tait, A., Henderson, R., Turner, R., Zheng, X. (2006). Thin plate smoothing spline interpolation of daily rainfall for New Zealand using a climatological rainfall surface. *International Journal of Climatology 26*, 2097-2115.

3 Impacts of climate change on soil natural capital and soil services

Bryan Stevenson[1], Nigel Bell[2], Saman Bowatte[2], Brent Clothier[3], Ian Dickie[1], Estelle Dominati[2], Maureen O'Callaghan[2], Kate Orwin[1], Roger Parfitt[1], Simeon Smaill[4], Steve Wakelin[2]

[1]Landcare Research, [2]AgResearch, [3]Plant & Food Research, [4]Scion

3.1 KEY MESSAGES

Table 3.1 provides a summary of the impacts of climate change on soil physical and biotic natural capital (the components that make our soils and landscapes). It takes into consideration soil processes and in particular the interaction of C and nutrient cycling, thus giving an estimation of sector relevance. Key messages are outlined below.

Soil physical natural capital and chemistry

- Little direct effect of eCO_2 on soil physical structure and chemistry is expected, but indirect effects will occur through changes to plant photosynthesis and organic matter turnover.
- A rise in atmospheric temperature will lead to warmer soil temperatures. Warmer soil temperatures will primarily lead to increased rates of chemical reactions, including those mediated by soil biota.
- Secondary impacts of increased temperature (from possible changes in soil C content) include changes to soil structure and macroporosity, soil water content and soil water repellency, and further changes to rates of N mineralisation.
- The changes to soil structure and soil water content will in turn affect soil aeration, redox reactions, GHG production and nitrate leaching.
- Changes in rainfall will lead to the primary impact of altered soil water content affecting reduction/oxidation (redox) reactions and leaching. Dryer soil will experience drought and induce soil water repellency, which enhances drought effects.
- Increased plant photosynthesis and greater C turnover could potentially lead to acidification and leaching of nutrients in unbuffered systems where pH is not controlled as a part of land management. There is some evidence that this has occurred in forests but not in pasture soils.
- Chemical reactions that occur relatively rapidly in soil (leaching of ions, change of pH, loss of silica (Si), redox reactions) will be particularly impacted by changes in temperature and moisture.
- If leaching of Si is accelerated in volcanic ash soils and South Island montane soils, this may lead to increased production of allophane and Al-humus complexes, possibly leading to increased P retention. The time scale for this is uncertain but is probably in the order of decades, if not centuries.

Soil biotic natural capital and relations to process and services

- Effects on soil microbial biomass are likely to be fundamental to the way soil responds to climate change, because soil biota drive C and nutrient cycling and have been described as the eye of the needle through which all nutrients must pass.
- Soil biota form a foodweb which is intricately linked with soil physical and chemical properties, as well as with plant communities. The complexity of feedbacks between

each component and the species-specific effects make it difficult to make general conclusions about the direct effects of climate change on soil biota.

- There is a major knowledge gap about how climate change will indirectly change feedbacks in plant behaviour, community structure of plants and biota, and soils.
- Pest and disease outbreaks are expected to increase and may negate some of the potential gains to plant production from eCO_2.
- Uncertainty about changes in biological N_2 fixation (BNF) in the legume-rhizobial symbiosis warrants further investigation given New Zealand’s reliance on white clover/Rhizobium symbiosis. There is strong evidence that eCO_2 will lead to increases in BNF in agricultural systems, but a reduced proportion of atmospheric N was fixed under eCO_2 conditions in the New Zealand pasture FACE experiment. It is not known, however, if this was due to nutrient limitation at this site.
- It is expected that eCO_2 will alter the bacterial:fungal ratio and increase saprotrophic biomass. The direction of change for the ratio is difficult to predict, but a decrease would be expected to be slow nutrient cycling and lower nutrient loss, and an increase to cause the opposite.
- Increased mycorrhizal biomass with eCO_2 appears to be a common (but by no means certain) outcome. Mycorrhizal fungi can enhance plant nutrient acquisition through increased competitiveness with soil microbes, and this may be important if progressive nutrient limitation (PNL) occurs. However, evidence suggests that increased mycorrhizal biomass may have both positive and negative effects on soil C, making their net impact difficult to judge.
- Because soil biota control C and nutrient cycling, they should be included in climate change models, but currently are not.
- Multiple resource limitation is a key factor limiting adaption to climate change. Stoichiometric calculations can help predict the affects of limited amounts of multiple nutrients.

Table 3.1: The impacts of climate change on soil processes considering the interactions between plants and soil organisms. The colours of the boxes represent which sectors are most likely to be impacted. ***There is a reasonable certainty of a change occurring in some sectors but there is high uncertainty in both the direction and magnitude of any changes.***

Factors potentially affected by climate change	Processes influenced by factors	Probability (by sector) that change to a factor will sufficiently affect a process to a relevant degree				Justification
		Cropping	Dairy	Extensive grazing	Forestry	
Species diversity or richness						
Plant	Organic matter decomposition	🟩	🟩	🟨	🟨	Greater potential for change in species diversity and less nutrient management in extensive grazing and forestry
	Nutrient cycling	🟩	🟩	🟨	🟨	
	Carbon dioxide and methane flux	🟩	🟩	🟨	🟨	
Soil biota	Organic matter decomposition	🟩	🟩	🟨	🟨	Species richness can influence decomposition rates in extensive grazing and forestry systems
	Nutrient cycling	🟨	🟨	🟨	🟨	Some nutrient pathways are narrow and there is potential for species diversity to significantly influence availability
	Carbon dioxide and methane flux	🟩	🟩	🟩	🟨	No change likely due to probable functional redundancy - except for importance of methanotrophic bacteria in forestry systems
Species Biomass						
Plant	Organic matter decomposition	🟨	🟨	🟥	🟥	Changes to biomass likely in all systems. Negative changes in the cropping and dairy sectors will be minimised by management practices
	Nutrient cycling	🟩	🟩	🟥	🟥	Any effect in the cropping and dairy sectors is likely to be overwhelmed by nutrient-management practices
	Carbon dioxide and methane flux	🟥	🟥	🟥	🟥	Likely changes across all sectors
Soil biota	Organic matter decomposition	🟥	🟥	🟥	🟥	Significant responses likely in all cases due to change in rates of activity with altered abundances
	Nutrient cycling	🟥	🟥	🟥	🟥	
	Carbon dioxide and methane flux	🟥	🟥	🟥	🟥	

Species Composition						
Plant	Organic matter decomposition	🟩	🟩	🟥	🟨	Greater potential for species composition change in forestry than cropping and dairy sectors and even more so in extensive grazing
	Nutrient cycling	🟩	🟩	🟥	🟥	Greater potential for species composition change in extensive grazing and forestry due to less nutrient management in these sectors
	Carbon dioxide and methane flux	🟩	🟩	🟥	🟨	Greater potential for species composition change in forestry than cropping and dairy sectors and even more so in extensive grazing
Soil biota	Organic matter decomposition	🟥	🟥	🟥	🟥	Important across all sectors due to influence on many soil functions
	Nutrient cycling	🟨	🟨	🟥	🟥	Some nutrient pathways are highly influenced by species composition, importance moderated by fertiliser use
	Carbon dioxide and methane flux	🟥	🟥	🟥	🟥	Important across all sectors due to regulation
Plant Behaviour (changes in productivity, carbon inputs, and litter quality)						
	Organic matter decomposition	🟥	🟥	🟥	🟥	Important across all sectors due to influence on many soil functions
	Nutrient cycling	🟩	🟩	🟥	🟥	Less nutrient management in extensive grazing and forestry
	Carbon dioxide and methane flux	🟥	🟥	🟥	🟥	Important across all sectors due to regulation

Direction of change and the probability that change to a factor will sufficiently affect a process to a relevant degree

🟥	Relatively high probability of change
🟨	Moderate probability of change
🟩	Low probability of change

3.2 INTRODUCTION

Soils are continually evolving and transforming within anthropogenic timescales (Richter et al., 2011) and these changes may positively or negatively affect the provision of soil services. Climate change is expected to accelerate soil change by altering temperature and precipitation, which are drivers of chemical and biological processes, and CO_2 levels, which affect plant photosynthetic efficiency (Tubiello et al., 2007). Therefore climate change will drive soil change directly by changing equilibria and the rates and frequency at which soil processes occur, and indirectly by influencing the soil physical structure and composition and stocks of plant communities and soil biota.

Although there is growing knowledge of soil physio-chemical and microbiological responses to climate change, the links between soil change and a change in the provision of soil services are still poorly understood because of the complex interactions behind each service. In particular, soil chemical and physical attributes interact with soil biota to sustain underlying biogeochemical processes such as C and nutrient cycling, which underpin soil services. Therefore, changes in soil natural capital (i.e. physical, chemical and biological soil properties or stocks) through time can either increase or decrease the level of services soils provide (Robinson et al., 2013). More understanding of these positive or negative shifts in soil properties and processes and the repercussion on the provision of services, including food production and GHGs emissions from soils (Singh et al., 2010; Bardgett et al., 2008; Young et al., 1998), is needed so that primary production can adapt to climate change in the mid- to long-term.

This section focusses on the impact of climate change on soil properties and processes, and thereby on the provision of a variety of regulating soil services such as C and nutrient cycling and GHG production. It is presented in five major parts:

3.3 Natural capital, ecological infrastructure, and ecosystem services.
3.4 Will climate change have relevant impacts on soil physical natural capital and processes?
3.5 Will climate change have relevant impacts on soil chemistry?
3.6 Will climate change have relevant impacts on soil biotic communities?
3.7 Will climate change have relevant impacts on soil-plant-microbe systems and feedback effects on soil services?

3.3 NATURAL CAPITAL, ECOLOGICAL INFRASTRUCTURE, AND ECOSYSTEM SERVICES

Soil has an inherent value in providing services to humans. The ecosystem services model provides a framework for valuing soil services (Costanza et al., 1997). Provisioning services are defined as the products obtained from soil (food and fibre) while regulating services enable humans to live in a stable, healthy and resilient environment (Table 3.2). This report focuses on how climate change will impact upon soil infrastructure which is the soil properties and processes that contribute to natural capital stocks (Bristow et al., 2012), and in turn provide **provisioning** and **regulating** services.

Provisioning services are usually associated with commodities in existing markets, so their value is readily apparent. Regulating services are often more difficult to put a monetary value on and are often overlooked in decision making. However, costs can occur if these services are compromised. Direct costs include damage caused by floods and erosion or loss of yield from drought or pests. Indirect costs include compensation to avoid loss of provisioning services, such as increased fertiliser use. Although cultural services are out of the scope of the review, they should be considered when taking ecosystem services as a whole into account.

Table 3.2: Provisioning and regulating ecosystem services provided by soil (Dominati et al., 2010).

Type of Service	Service delivered	Description
Provisioning services	Provision of food, wood and fibre and products	Soil physically supports plants and supplies them with nutrients and water. A wide range of plants are grown by humans and harvested for a variety of purposes.
	Provision of raw materials[1]	Soil can be source of raw materials such as peat and clay.
	Provision of support for human infrastructure and animals.	Soil represents the physical base on which human infrastructures and animals stand.
Regulating services	Flood mitigation[2]	Soil has the capacity to store and retain water, thereby mitigating flooding.
	Nutrient and contaminant filtration	Soil can absorb and retain nutrients and contaminants, which prevents them from being released into water bodies.
	Carbon storage and greenhouse gases regulation	Soil can store carbon and regulate the production of greenhouse gases.
	Detoxification and the recycling of wastes	Harmful compounds can be physically absorbed by soil or destroyed by organisms that exist in soil. These organisms also degrade dead organic matter, which improves soil structure and releases nutrients.
	Regulation of pest and disease populations	The nature of the habitat provided by soil controls the proliferation of pests (crops, animals or humans) and harmful disease vectors (viruses, bacteria), and regulates populations of beneficial species.

Soil natural capital stocks are the physical, chemical and biological properties that make up New Zealand’s landscapes and productive sectors. Some properties can be managed, such as altering levels of nutrients by adding fertiliser. Soils are also dynamic systems with various degradation and supporting processes occurring continuously. These processes along with natural capital form the ecological infrastructure of soil (Figure 3.1). The relationships between soil natural capital and the flow of ecosystem services are dependent on the complex interaction between natural capital stocks and soil processes. Behind each service, a number of soil natural capital stocks can be regulated by multiple soil processes and each soil process may, in turn, contribute to several stocks and services (Robinson et al., 2013).

Zhang et al. (2007) present a framework that links the non-market services from agricultural ecosystems to receipt of regulating services. They also note that there are ecosystems disservices to agriculture as a result of pest damage, competition for pollination, and competition for water. As well, agriculture provides ecosystem disservices through habitat loss, nutrient runoff, and non-target pesticide impact. Their assessment shows that the impacts of climate change need to be considered from the perspective of both service and dis-service. In essence, this distinction can be seen in Figure 3.1, where Dominati et al. (2010) consider that climate change can result in changes to the soil’s properties that can either enhance supporting processes, or engender degradation processes that reduce the soil’s natural capital value by diminishing ecosystem services.

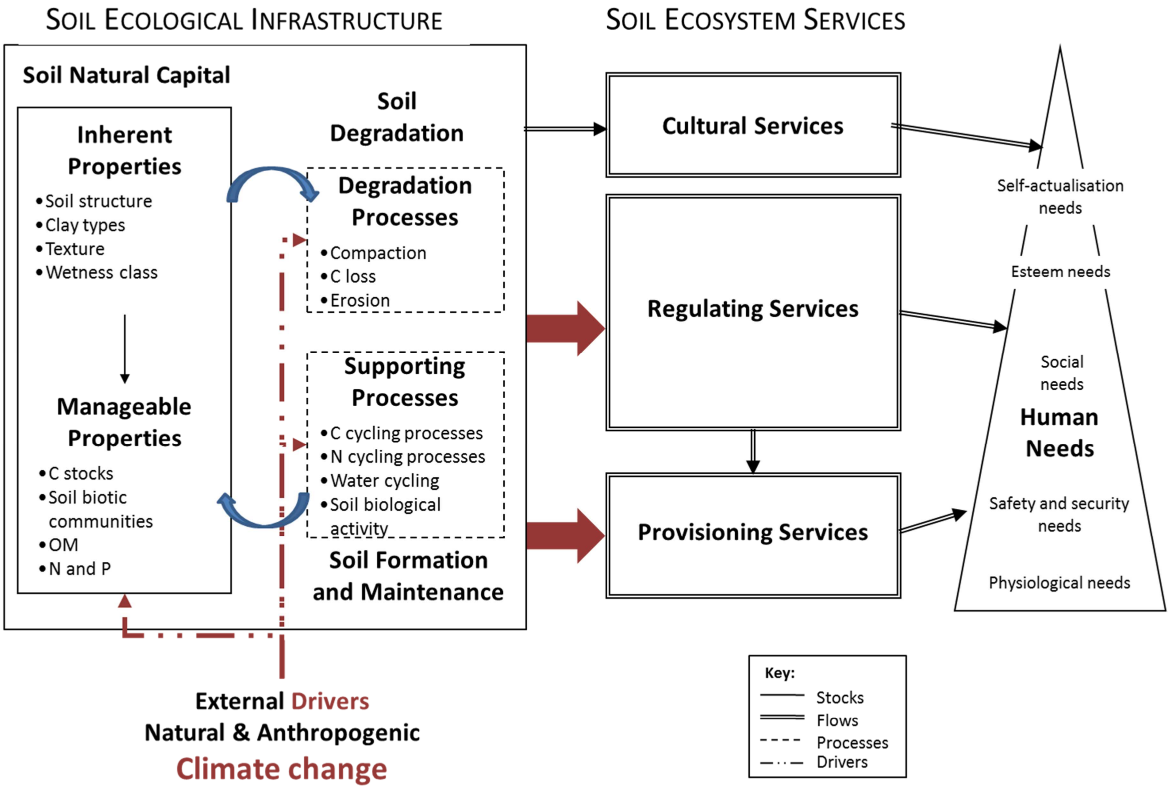

Figure 3.1: Conceptual diagram of relationship between climate change drivers, soil natural capital, soil processes and ecosystem services (adapted from Dominati et al., 2010).

The approach taken in this review focuses on the mechanisms underlying the provision of services that will be affected by climate change. The delivery of soil regulating services and the vulnerability of the provisioning services to climate change are assessed by considering the underlying soil natural capital stocks and processes.

3.4 WILL CLIMATE CHANGE HAVE RELEVANT IMPACTS ON SOIL PHYSICAL NATURAL CAPITAL AND PROCESSES?

The physical natural capital of soils is the arrangement of solid particles, water and air that comprise soil structure. The mineralogy and particle size of the parent material is a major influence on soil structure but other soil properties can modify structure, particularly C.

The focus here will largely be on the impact of climate change on supporting processes (carbon, nitrogen and water cycling processes, and soil biological activity) and regulating services. The regulating services we will consider here are shown in the context of the mechanisms of changes driven by climate change.

The results of climate change we will consider here are changes in air temperature and changes in rainfall. We consider how these will affect the soil's natural capital properties of soil temperature, soil water content, soil carbon, macroporosity, biological activity, and hydrophobicity or soil water repellency (Figure 3.2). The changes in these soil properties have an impact on the delivery of water, gas and nutrient regulation services.

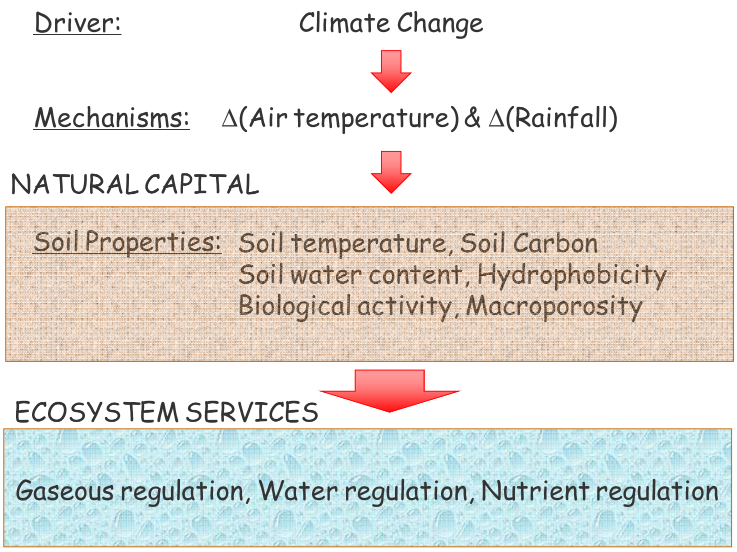

Figure 3.2: The mechanisms of climate change (temperature and rainfall), and the impact such changes will have on the soil's delivery of the ecosystem services of water, gas and nutrient regulation.

We will not consider the direct impact of changed levels of CO_2 on the delivery of soil ecosystem services because we believe the changes in CO_2 will not greatly change the soil's physical properties.

The recent book by Kirkham (2011) has extensive bibliographies and provides an up-to-date and comprehensive review of our current knowledge on the impacts of eCO_2 on soil biophysical properties and plant-water relations. The majority of effects described in this book come from changes in plant physiological properties rather than the direct impact of eCO_2 on the soil's physical properties.

Increased air temperatures

Increased air temperatures will increase soil temperatures, which will impact on soil biological activity and the soil's C content. These changes could have significant impacts, both within the soil and well beyond.

Soil C content & CO_2 regulation

In a significant paper in *Nature*, Cox et al. (2000) predicted that, with the global rise in CO_2 and temperature, the altered balance between the vegetation sink and increased soil respiration would mean the land would turn from being a net C sink to a net emitter by 2050. Luke and Cox (2011) referred to this increased respiration of soil C and atmospheric feedback as the potential runaway influence of temperature on soil respiration, or the 'compost-bomb instability'. They modelled the link between global warming, soil temperature, soil respiration, and soil C according to the links and feedback shown in Figure 3.3.

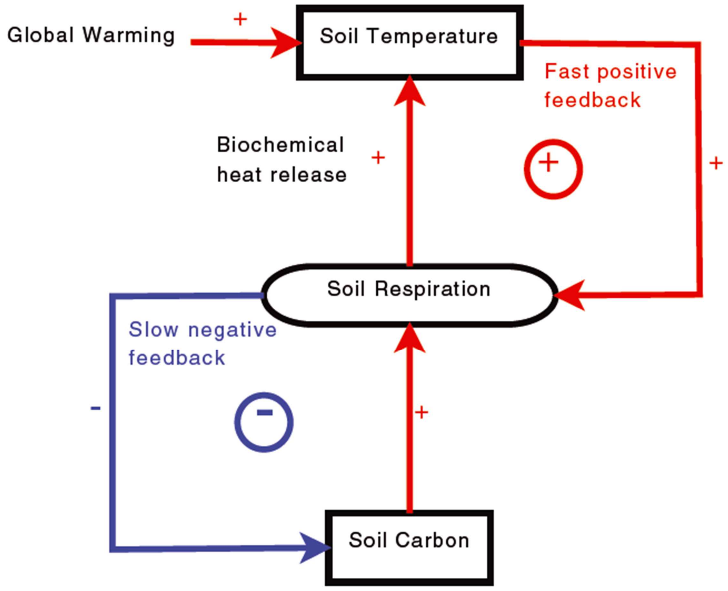

Figure 3.3: The links and feedback between global warming, soil temperature, soil respiration and soil carbon modelled by Luke and Cox (2011).

The criterion for this instability depends on three things: the slope of the temperature response of Gross Primary Production, the q_{10} for soil C respiration, and surface temperature response to a doubling in CO_2 (Luke & Cox, 2011). The q_{10} value is the change in reaction rate following a 10˚C temperature rise. For the compost-bomb to ‘explode’, they predicted the global air temperature rise would need to be 10˚C per century (Figure 3.4), which is greater than current predictions. Nonetheless, increased decomposition of soil C from increased temperature is a concern (Figure 3.4). Although New Zealand’s temperature increase in is projected to be lower than global increases, general projections are that soil C levels are likely either to stay the same or decline (see discussion in section 4). Any decline in soil C will also affect nutrient and water regulation services.

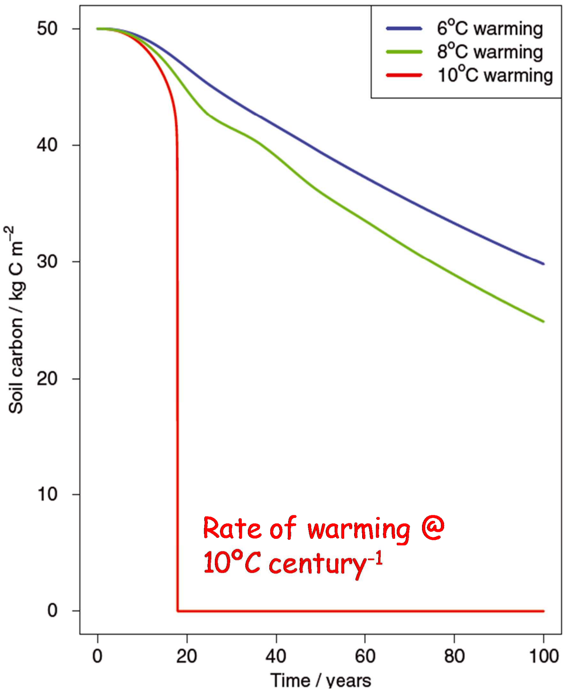

Figure 3.4: The rate of rise in global warming (˚C per century) and predicted time course of changes in the stocks of soil C (Luke & Cox, 2011)

Nutrient regulation

The effects of soil C management, environmental conditions and the role of soil structure on N mineralisation in orchard soils were examined by Kim et al. (2011). They showed that warmer temperatures, combined with soils drier than saturation, will lead to greater N mineralisation. But the dominant effect, some 50% of the variability, was due to the soil's hot water C content (Figure 3.5). Thus, should global warming decrease the soil's C content, and in particular its labile, hot-water C content, the nutrient regulating service provided naturally by the soil will be degraded. Conversely, if there is investment of C through agricultural practices that increase the soil's C content, particularly its hot water C content, there will be increased nutrient regulation.

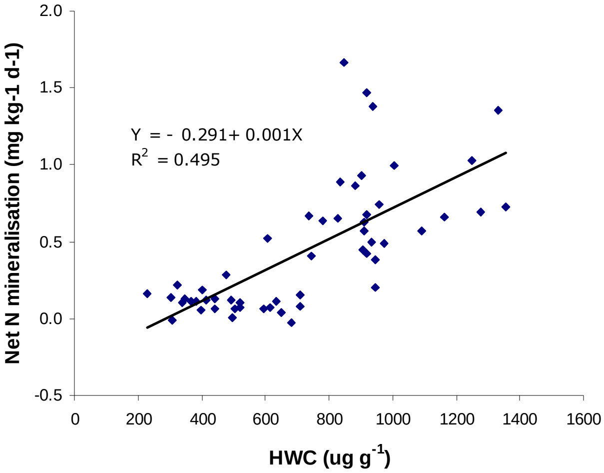

Figure 3.5: The correlation between the soil's hot water carbon content (HWC) and N mineralisation (Kim et al., 2008).

Water regulation services

Many studies have aimed to determine the impact of soil C on the soil's ability to store and buffer water. Contradictory findings have been reported, but there is general agreement that soil texture affects both soil C and soil water content. Rawls et al. (2003) hypothesised that the effect of soil C on water retention would depend on both the textural make-up of the soil and the level of soil organic matter itself. To test this they used the comprehensive U.S. National Soil Characterization Database. The impact of increasing soil organic C is greatest for soils with low initial soil C. There are large and positive increases at low clay contents. This highlights the benefit of using agricultural practices to maintain and enhance the soil's C content in such soils, for it would provide better water regulation services. However, increasing organic C caused negative changes in the soil-water buffering capacity in soils with high clay contents. In soils with high organic matter contents, the water regulation services through increasing soil C contents through soil management were always enhanced, although of a lesser order. The framework of Rawls et al. (2003) could also be used to consider what impacts the rise in air temperatures may have on water regulation services by considering the changes in soil C predicted by models.

Gaseous regulation services

Using X-ray tomography, Deurer et al. (2009) showed how different soil C management practices between two neighbouring apple orchards affected the soil's macropore (large soil pores) structure. One orchard had 3.8 kg C m^{-2} and the other just 2.6 kg C m^{-2}. The high C soil's connected macroporosity was dramatically higher (Figure 3.6). This indicates greater gaseous diffusion, which would indicate less favourable conditions for nitrous oxide (N_2O) production and emission. This would provide a beneficial regulating service for the atmosphere in terms of a reduction in atmospheric greenhouse gases.

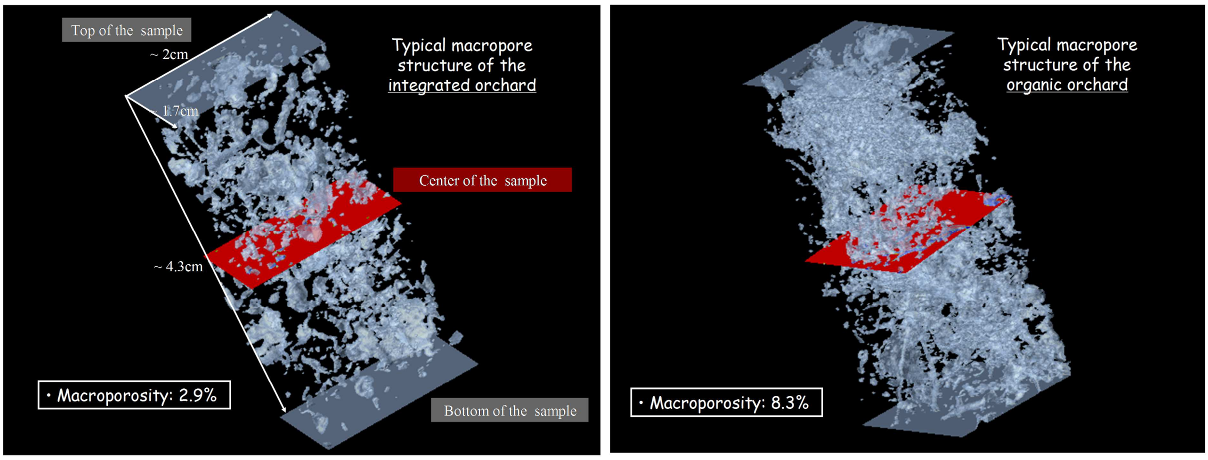

Figure 3.6. X-ray tomographic images of two identical soils that have undergone different C investment strategies, resulting in a different ecological infrastructure with relation to macroporosity. The soil on the left has 2.6 kg C m^{-2} and the soil on the right has 3.8 kg C m^{-2}.

The results of van der Weerden et al. (2012) confirm this link between macroporosity, diffusion, and N_2O emission. They found that in a New Zealand pasture soil, increased pore continuity shortened the duration of anaerobicity, leading to lower emissions. Indeed extrapolating from the regression equations for their two soils, would suggest that N_2O emissions from the high-C soil would be 15–35 times lower than those in the low-C soil (Figure 3.7).

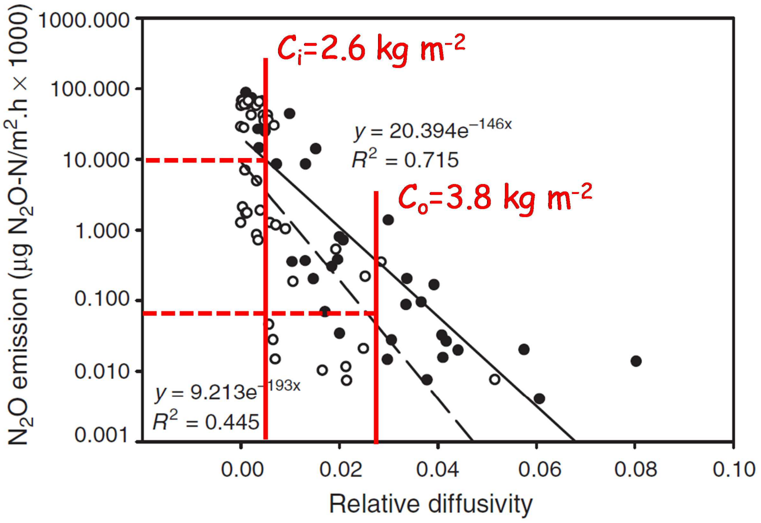

Figure 3.7: Relative diffusivities for the two soils with different carbon levels (C_i and C_o) from Figure 3.6, in relation to the nitrous oxide emissions measured by van der Weerden et al. (2012).

Thus increases in soil C, whether by global warming or agricultural practices, will alter the soil natural capital value by increase the soil's macroporosity. This will improve the gaseous regulating services delivered by soil.

Soil C, macroporosity and nutrient regulation

Soil macropores are a manageable soil property that is likely to be affected by climate change. Macroporous networks are maintained and enhanced by sustaining soil C levels, as described by Deurer et al. (2009). Depending on whether the C is exogenously applied or endogenously generated, these bypass networks can either provide a service or disservice. They can provide a valuable nutrient regulation service by limiting leaching losses (Green et al., 2010), or a disservice by enhancing the preferential loss of nutrients via leaching (Cichota et al., 2010).

An example of limiting leached losses was provided by Green et al. (2010). This study found only 8–13% of the endogenously generated N as a result of N mineralisation in the soil's matrix was wastefully leached below the roots and into the vadose zone. Rain was shown to fall through the macropores, thereby avoiding contact with the N mineralised within the soil's matrix.

Exogenous N in the form of urine patches was shown to enhance nutrient loss by Cichota et al. (2010). Some 45–65% of the applied N was lost to the soil-plant system after being picked up by rainfall and lost through the macropores.

A rudimentary calculation was made by Clothier et al. (2008) suggesting that the global value of the ecosystem services provided by macropores in soil was US$304 billion per year. Investing C into the soil therefore has a huge value though its boost to macroporosity. First, this investment sequesters C in the soil, preventing loss to the atmosphere, and also enhances N mineralisation, and limits N_2O emissions and the leaching of the endogenously mineralised N. Nonetheless, enhanced macroporosity does generate a disservice in relation to regulating exogenously applied nutrients.

Soil water storage, drought, and water regulation

The effects of climate change on drought, soil water demand, soil water storage and feedbacks to water regulation are intricantly linked. Climate change will likely bring increased demand for irrigation created by higher temperatures and changed rainfall patterns. The soil water regulating service could be changed and require greater use of irrigation. Likewise, the way soil controls runoff to surface waters and drainage recharge to groundwaters will also be affected by climate. Here we report some recently published findings on the impacts on, and adaptation options for, future horticulture in New Zealand (Clothier et al., 2012).

The modelling framework of SPASMO (Soil Plant Atmosphere System Model) was used to assess irrigation needs and the impact of horticulture on groundwater quantity and quality (Clothier et al., 2012). The high carbon world (A2) and rapidly decarbonising world (B1) climate change scenarios in 2050 were used to assess a wide range of impacts, including irrigation. Modelling of future irrigation requirements was carried out for apples in Central Hawke's Bay, kiwifruit in the Bay of Plenty, and grapes in Marlborough. The irrigation requirements, now and into the future, were calculated for soils with different water-holding capacities, or different buffering potentials against drought.

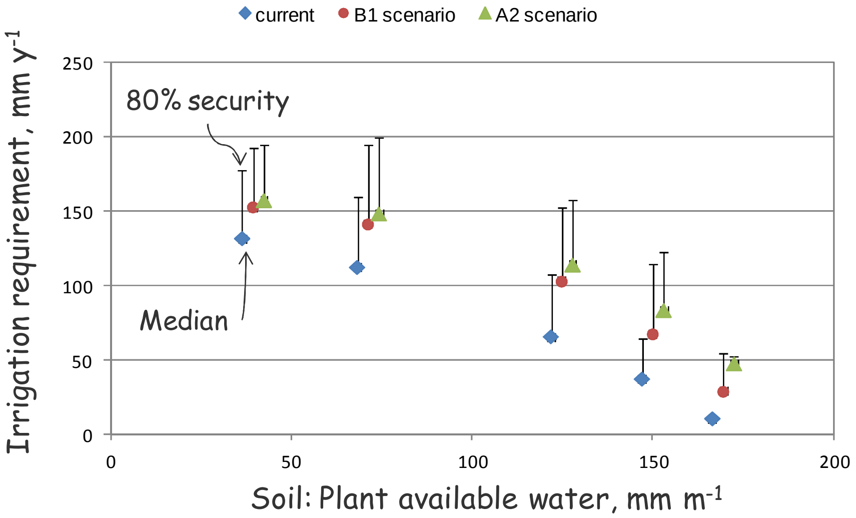

Figure 3.8: The requirement for irrigation water in 2050 for grapes growing in Marlborough on soils with different water holding capacities. The symbols are for the median water requirement (50% security) and the upper bar is the requirement that will meet needs 8 years out of 10 (from Clothier et al., 2012).

The results showed little increase in water demand for central Hawke’s Bay, but almost double the need in the Bay of Plenty. In Marlborough (Figure 3.8), water needs were increased, with an increase of between 20 and 50 mm of irrigation water per year required to meet demands that will not be supplied via the soil’s water regulating service. The study also showed that groundwater recharge by drainage through the rootzones of vineyard soils in Marlborough would be 5–8% less by 2050. In Marlborough there will be future pressures on water as a result of climate change, and rainfall and the soil’s water regulating service will not provide sufficient buffering for viticultural water requirements.

Hydrophobicity – a degradation process?

Soil water repellency (SWR) is when a soil does not wet up spontaneously when water is applied to it, and predicted decreases in rainfall in some areas under climate change mean that it may become more persistent. The contact angle between the water and the soil surface determines the degree of hydrophobicity. A high contact angle (>90˚) indicates hydrophobicity. The phenomenon of SWR can be considered as a disservice (Zhang et al. (2007)).

In New Zealand, Müller et al. (2010b) found SWR reduced infiltration rates by up to a factor of 20 on a Waikato farm, and pasture growth was reduced between 5 and 20%. Jeyakumar et al. (2012) reported surveys across both islands of New Zealand showing that the majority of soils showed potential SWR characteristics. They also carried out laboratory and field-scale (simulated) rainfall and runoff experiments. The laboratory tests showed large-scale runoff under SWR, but the field experiments did not because surface-vented macropores and cracks readily captured the runoff, further highlighting the service value of macropores. Nonetheless, ‘dry patches’ were created (see inset in Figure 3.9) where the water first ran off, accounting for the loss in pasture production measured there. Müller et al. (2010b) calculated that this ‘dry patch syndrome’ results in a loss of about 30–40% in pasture production.

Failure by SWR of the soil’s water-regulating service of infiltration leads to a loss in the provision service of pasture growth.

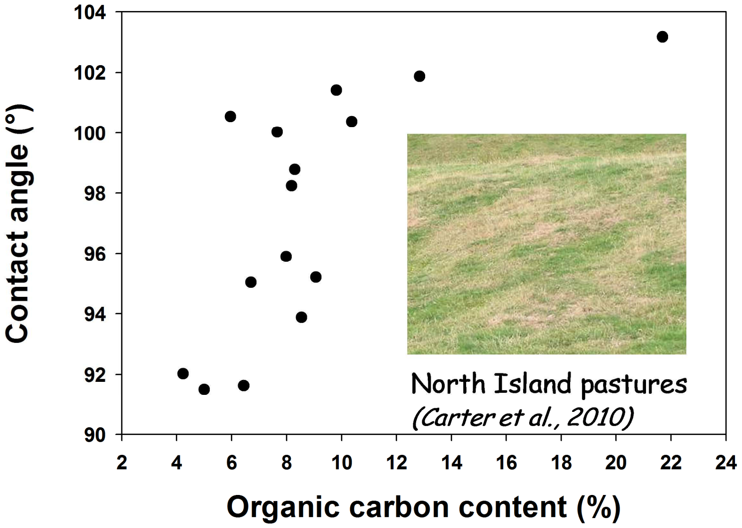

Figure 3.9: The plot shows the contact angle as a function of soil C content. When the contact angle exceeds 90˚ the soil is water repellent. Inset: Dry patch syndrome, a phenomenon caused by soil water repellency, which limits pasture growth and enhances runoff.

The phenomenon of SWR occurs when the soil surface's water content drops below a critical soil water threshold (CWT) value. This is likely to occur in late spring and early summer, and not be relieved until autumn (Figure 3.10). During the intervening summer period, the soil's water regulating service is degraded because of SWR.

However, not all the specific impacts of SWR are a disservice. Robinson et al. (2010) used this concept to understand the ecosystem service played by SWR in piñon-juniper woodlands. The trees induced SWR resulting in greater runoff which was captured by cracks that funnelled the water deep into the soil profile. The water was protected from surface evaporation, and was only available to the piñon and juniper trees, rather than the understorey of the shallow-rooted herbaceous plants. This strategy might be of benefit for the trees and vines of New Zealand's horticultural systems.

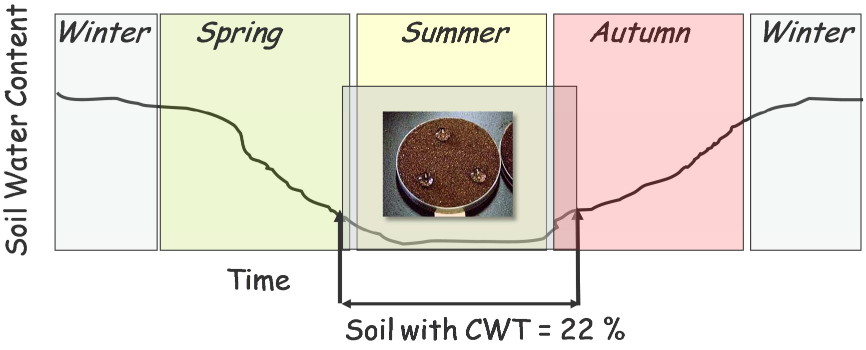

Figure 3.10: A hypothetical plot of the soil's water content throughout the year, highlighting the period when the water content drops below the Critical Water Threshold value (CWT, here say 22%) when the soil exhibits hydrophobicity, or soil water repellency.

Müller et al. (2010a) tested whether there was a link between SWR, eCO_2 and soil C mineralisation at the NZFACE site in the Manawatu. There are no other known reports on CO_2 levels and SWR. No significant differences in the persistence or degree of SWR were found, and eCO_2 had no effect on C mineralisation. They concluded that SWR does not contribute to increasing the long-term terrestrial sink in response to eCO_2.

Synthesis and summry of impacts of climate change on soil physical natural capital

In this subsection we have explored the impact of climate change on the ecosystem services that flow from those physical properties of the soil's natural capital. Our focus is on the soil's physical properties, so we have limited our assessments to altered temperature and rainfall patterns. These changes are summarised in Figures 3.11 and 3.12 along with critical feedbacks and inter-linkages.

Temperature

A rise in air temperature will lead to warmer soil temperatures, primarily leading to changes in the soil's physical properties and processes, notably changed rates of N mineralisation and possible changes in the soil's C content (Figure 3.11). Any change in the soil's C content will eventually lead to changes in the atmosphere's CO_2 levels which will further impact on air temperatures (Figures 3.3 and 3.4).

Changes in the soil's C content will have secondary impasts on soil macroporosity, N mineralisation and soil water content and macroporosity (Figure 3.11 and 3.12).

Moving to the next level in Figure 3.11, macroporosity will have a tertiary impact on soil aeration (Figure 3.6), nitrous oxide emission (Figure 3.7) and nitrate leaching (see Section on Nutrient Regulation; and Robinson et al., 2013).

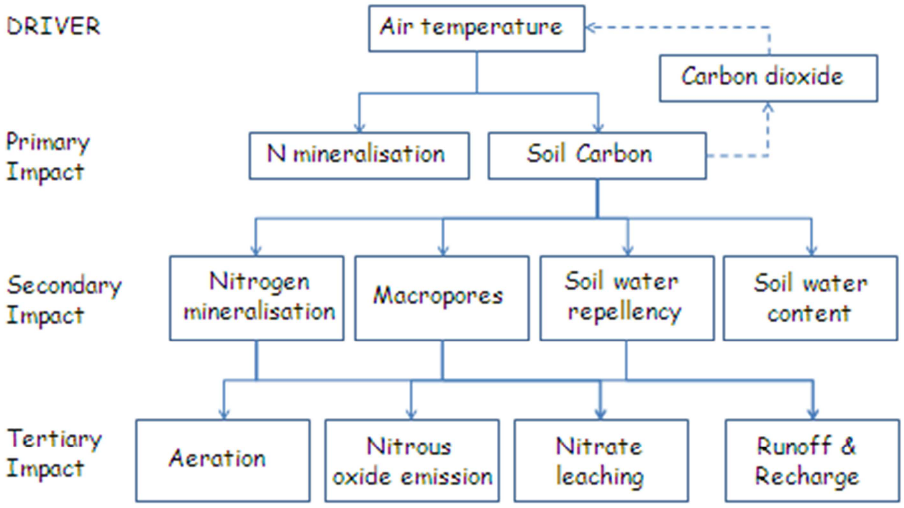

Figure 3.11: The link between air temperature and its impacts on the soil's physical properties and the soil ecosystem services that flow from them. Solid lines indicate direct effects, and dashed lines indicate feedback cycles.

The changed patterns of soil water content and soil water repellency induced by the secondary impact of changes in soil carbon will also affect rainfall runoff and groundwater recharge (Figures 3.12).

Rainfall

Changed patterns of rainfall as a result of climate change will have their primary impact on the soil's water content.

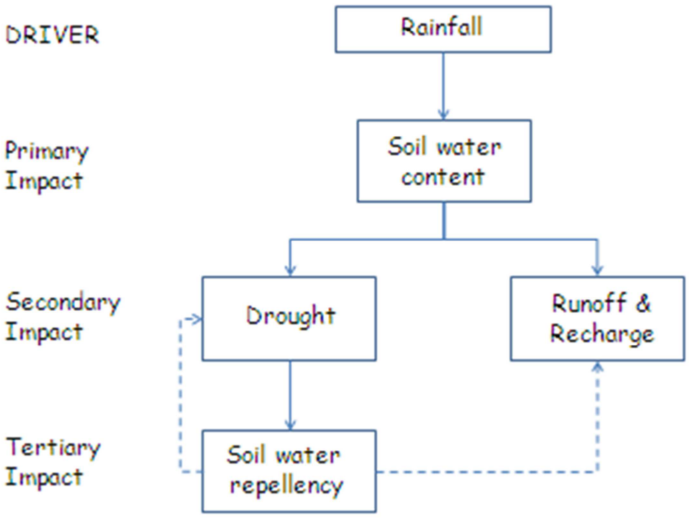

Figure 3.12: The link between rainfall and its impacts on the soil's physical properties and the soil ecosystem services that flow from them. Solid lines indicate direct effects, and dashed lines indicate feedback cycles.

Altered soil water content will lead to altered patterns of drought and changed patterns in the ecosystem service of runoff and recharge (Clothier et al., 2012). Drought will affect the provision service from the soil, unless irrigation water is available to offset the lack of rainfall.

The changed patterns in drought will have a tertiary impact on soil water repellency (SWR) because the timing when the soil-water content drops below the CWT (Critical Water Threshold) will change. If SWR increases, this will exacerbate drought, hence the feedback in Figure 3.12, as neither the occasional summer rains nor the drought-breaking autumnal rains will be as effective at rewetting the soil (Figures 3.9 and 3.10). As a consequence of this ineffective wetting, the rainfall runoff and groundwater recharge from the soil will also be affected. Note that SWR is affected by the soil's C content (Figure 3.11), and that SWR appears as an impact in both Figure 3.11 in relation to temperature, and in Figure 3.12 due to rainfall.

Figures 3.11 and 3.12 highlight the linkages between temperature and rainfall and the hierarchy of impacts that these will have on the delivery of ecosystem services that depend on the soil's physical properties. The lynchpin in this delivery is the soil's C content. Adaptation and mitigation options are nonetheless available, as soil carbon levels can be maintained by good agricultural practices to enable the delivery of appropriate ecosystem services in the face of changed air temperatures and rainfall patterns.

3.5 WILL CLIMATE CHANGE HAVE RELEVANT IMPACTS ON SOIL CHEMISTRY?

The interaction of solid particles (soil minerals and organic complexes) with soil water and air also affects chemical reactions in the soil. The effect of climate change on soil chemical processes is not only dependent on the direct effects of climate change, but the indirect effects through changes in vegetation, soil physical structure and biota. Soil chemistry is also intricantly linked with soil management as pH in particular is often regulated in more intensively managed systems.

Brinkman and Sombroek (1996) state that "In most cases, changes in soils by direct human action, on-site or off-site (whether intentional or unintended), are far greater than the direct climate-induced effects. Soil management measures designed to optimize the soil's sustained productive capacity would therefore be generally adequate to counteract any degradation of agricultural land by climate change. Soils of nature areas, or other land with a low intensity of management such as semi-natural forests used for extraction of wood and other products, are less readily protected against the effects of climate change but such soils, too, are threatened less by climate change than by human actions – off-site, such as pollution by acid deposition, or on-site, such as excessive nutrient extraction under very low-input agriculture". With the effects of management on soil chemistry in mind, aspects of the soil system that are likely to change within the next century are reviewed.

The most likely changes in soil-forming factors resulting from climate change are in organic matter supply (an indirect effect on changes to the plant community), soil temperature regime and soil hydrology, the latter because of changed rainfall events as well as changes in potential evapotranspiration. It is assumed that temperature will increase, but changes in rainfall patterns are uncertain.

Atlhough there is evidence that large amounts of CO_2 can acidify soils and increase weathering (Stephens and Hering, 2002, 2004), these data come from studies of volcanic rocks near volcanic vents with very high concentrations of CO_2. The direct effecs of rising atmospheric CO_2 concentrations (365–600 $\mu l\ l^{-1}$) will be much less concentrated, and may cause a slight decrease in soil pH; this is likely to be small in comparison with other causes of pH drop.

Changes to soil hydrology resuting in redox conditions can be rapid because soils can become anaerobic when the water table is raised and oxygen is depleted. A process called ferrolysis – the cyclic transformation and dissolution of clays triggered by alternating iron reduction and oxidation – can cause a decrease in the cation exchange capacity by aluminium interlayering in clay minerals (Brinkman, 1982). This effect, however, is more related to rainfall and drainage than to temperature. Increased temperature, however, may lead to a reddening of presently brown soils if increased periods with high summer temperatures coincide with dry conditions, so that the iron oxide haematite would be formed rather than ferrihydrite and goethite. This may lower the phosphorus (P) retention of these soils because hematite is less reactive to P. This change is unlikely to be noticeable over a period of about 50 years, but may occur in the longer term.

Increased temperature will likely increase the rates of weathering of minerals in soils, but the clay mineral composition and the mineralogy of the coarser fractions would probably change little over hundreds of years. Changes in the surface properties of the clay fraction can take place faster, however. For instance, studies of weathering of granitoid rocks suggest that temperature (in the range of 5–35°C) can alter natural silicate weathering rates. Release of potassium (K) concentrations may become elevated with respect to other cations due to the rapid oxidation/dissolution of biotite (White et al., 1999). In a study of 68 watersheds underlain by granitoid rock types, Na weathering fluxes showed increases with precipitation, runoff, and temperature. A model that gave a prediction of weathering rates over climatic extremes indicated that fluxes of Si, Ca, and Mg exhibited no climatic correlation, implying that other processes, such as ion exchange, nutrient cycling and variations in lithology obscured any climatic signal. The correlation between yearly variations in precipitation and solute fluxes within individual watersheds was stronger than the correlation between precipitation and solute fluxes of watersheds with different climatic regimes. This correlation shows the importance of transport-induced variability in controlling chemistry, and the importance of distinguishing between short-term and long-term climatic trends (White & Blum 1995).

In a review on the nature of the links between weathering and the C cycle it was concluded that more work was needed on the relationships between weathering and erosion (Goudie & Viles, 2012). This is important to climate change models because of the link between silicate weathering and $C0_2$: hydrolytic weathering of silicate minerals may consume carbonic acid and thereby remove atmospheric CO^2 more rapidly with increasing temperature (Velbel, 1993). Indeed, a critical uncertainty in models of the global C cycle and climate is the combined effect of organic activity, temperature, and atmospheric CO_2 on silicate weathering.

Studies of dissolution rates of anorthite and augite indicate that silicate weathering in organic-rich solutions is not directly affected by soil CO_2 but is very sensitive to temperature. Apparently eCO_2 may accelerate silicate weathering indirectly by increasing organic activity and the production of organic acids (Brady & Carroll, 1994).

These results were generally confirmed for forest soils in the Duke Forest FACE experiment where eCO_2 concentration (+200 ppmv atmospheric) showed that increased soil CO_2 accelerated the rates of soil acidification and mineral weathering. An increase of 55% in atmospheric CO_2 concentration over 2 years resulted in a 271% increase in soil solution cation concentration, a 162% increase in alkalinity and a 25% increase in Si concentration at 200-cm depth. The flux of dissolved inorganic C to groundwater increased by 33%, indicating a negative feedback to changes in atmospheric CO_2 that could regulate the global C cycle over geological time. These changes to soil CO_2 dynamics were most likely the result of increased root and rhizosphere respiration, as suggested by the changes to the delta ^{13}C of soil CO_2 (Andrews & Schlesinger, 2001).

The ability of the soil to sorb positively charged nutrients is largely determined by the soils cation exchange capacity (CEC). CEC in turn, is partially determined by the C content of the soil. Thus, any declines in soil C will negatively affect the soil's ability to retain any increase in nutrients from weathering.

If leaching of Si is accelerated in volcanic ash soils and South Island montane soils, this may lead to increased production of allophane and Al-humus complexes (Parfitt, 2009), possibly leading to increased P retention. The time scale for this to occur is uncertain. If leaching of cations also increases, then in pasture soils the losses may include Ca, because bicarbonate is a dominant ion (produced by respiration in hill country) and with the charge balanced by Ca. In New Zealand forest soils Na and Cl tend to be the dominant ions, but these arise from wind-blown salts (Parfitt et al., 1999); the losses of other cations in forest soils will depend on the soil parent minerals and their mineralogy.

In summary, changes to soil chemistry from climate change are expected to be relatively minor in the near-term in comparison to other human impacts (such as land use change).

3.6 WILL CLIMATE CHANGE HAVE RELEVANT IMPACTS ON SOIL BIOTIC COMMUNITIES?

The many organisms that inhabit soil are diverse and interact in complex foodwebs to strongly influence soil processes. Perturbations to soil that disrupt the biodiversity and/or functioning of the soil biota can have profound influences on the services humans gain from soil. Regulating services outlined by Dominati et al. (2010), in which soil biota are involved, include nutrient recycling and retention, degradation of organic matter, regulation of pest and disease populations, regulation of greenhouse gas emissions, and degradation of harmful compounds. The degree to which the five major changes to existing climate (vis: eCO_2, elevated air temperature, changes in precipitation pattern and amount, interactions of climate change factors, and extreme weather events) impact on soil biota are discussed in this section.

The diversity of soil organism size from the micro- to macro-scale (Figure 3.13), along with the diversity in biology and ecology of these organisms, mean the impacts of climate changes vary across organism groups.

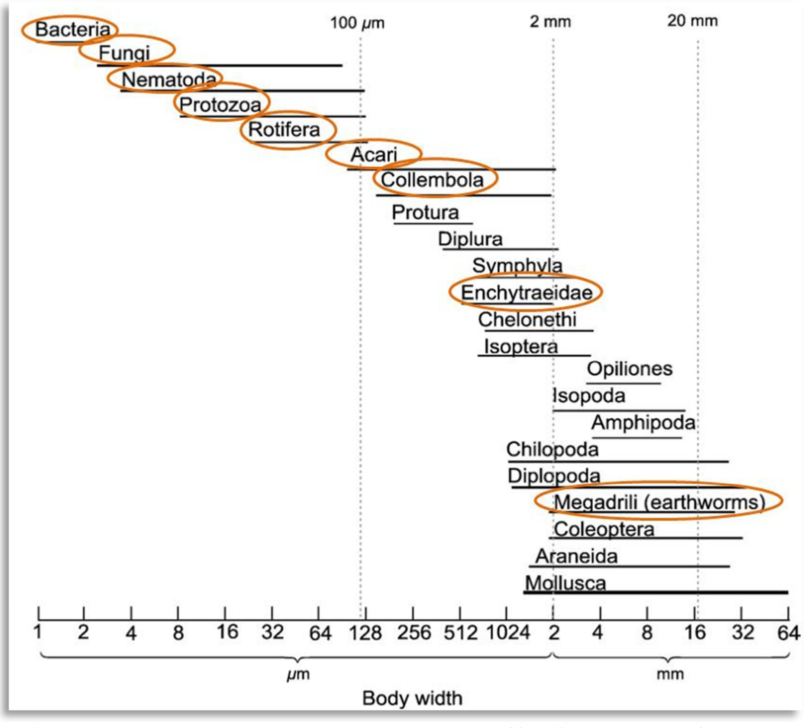

Figure 3.13: Size of soil organisms according to body width (Swift et al., 1979), with groups included in this review circled. On this scale microfauna are <100 μm; mesofauna 100 μm to 2 mm; macrofauna 2–20 mm and megafauna >20 mm.

Body size of soil fauna is useful not only as a classifier but also indicates the soil habitats they occupy. Habitats may respond differently to climate change factors, for example, those invertebrates that occupy soil pore water spaces may be more adversely affected by drying conditions than those that occupy soil aggregates. In general, the smaller the organism, the less we know about them, with less than 10% of the estimated total of fungi, bacteria, nematodes, protozoa and mites currently described, compared with more than 50% of soil insects (Wall et al., 2001). Nematodes appear often in the climate change literature that deals with micro- and mesofauna, which is unsurprising given they are the most numerous animals on earth and have a diverse range of feeding habits that makes them amenable for use as indicators of changes in soil condition (Wilson & Kakouli-Duarte, 2009). Of the macrofauna, earthworms receive attention because of their important role in soil process.

Species diversity and resolution within groups add uncertainty to attempts to summarise likely changes in soil biota communities under various climate change scenarios. A further source of uncertainty is the often conflicting results from studies conducted in different countries and different soils. In most cases, there are insufficient studies conducted in comparable situations (e.g., grazing intensity, plant species composition, and climatic zone), and at similar scales, to draw definitive conclusions.

In order to make this review targeted and relevant, we will largely limit this review to agricultural systems, most particularly pastures and forests, as these systems account for a large proportion of the area of managed landscape in New Zealand, making them good targets for any future mitigation and adaptation options. Previous reviews of impacts of climate change on soil biota deal with microbes (Beed et al., 2011) and invertebrates (Cock et al., 2011), but many of these have a northern hemisphere or non-agricultural focus. Where data are sparse for agricultural systems or particular organism groups, we have extended the review to studies in other ecosystems, particularly where these have been conducted under conditions which exist in at least some New Zealand agricultural situations.

In general, elevated above-ground CO_2 levels are unlikely to have a direct effect on soil biota given the much greater levels and fluctuations that exist in soil pores and pore water. Flechard et al. (2007) and Maier et al. (2010) measured soil pore CO_2 levels of >3500 ppm at 7–10 cm depth at some times of year beneath cool, temperate European pasture (ungrazed) and forest, respectively. Order of magnitude greater CO_2 concentrations were observed at greater depths, with up to 50,000 ppm occurring at 50 cm depth (Flechard et al., 2007). Diurnal fluctuations of 100–500 ppm were also observed (Flechard et al., 2007; Maier et al., 2010). Soils in those studies had pHs in the range 6–8 (Flechard et al., 2007) and 7–8 (Maier et al., 2010), which are higher than many New Zealand agricultural and forestry soils. Despite these differences in pH it is likely similarly high levels of soil pore CO_2 exist in New Zealand soils, so predicted above-ground rises in CO_2 levels from the current approximate 350 ppm to as much as 700 ppm are unlikely to directly impact soil biota.

Indirect effects of eCO_2 on soil biota are likely to be mediated via changes in plant community composition, production and subsequent litter quality and quantity. The effects of eCO_2 on these plant factors have received some research attention in pastures (e.g., Allard et al., 2004, 2005, 2006; Izaurralde et al., 2011; Parsons et al., 2011) and some of the subsequent effects on soil biota have been revealed.

Soil invertabrates (macro- and mesofauna)

Soil invertebrates include arthropods, nematodes and earthworms. We discuss some general climate change impacts on soil invertebrates, but largely focus on nematodes and earthworms. Nematodes are a family of small worms. Soil invertebrates can either feed on organic matter and bacteria (decomposers) or live plant material (grazers). Some groups of nematodes in particular are grazers, and are considered pests because they feed on plant parts (primarily roots) and can decrease plant yield. Earthworms feed on dead organic material and are important for a healthy soil as they mix organic matter and nutrients in the soil as they burrow.

In New Zealand, increased populations of fungal feeding, omnivorous, and predacious nematodes, along with enchytraeid worms and earthworms, were observed in pasture soil subjected to eCO_2 (700 vs 350 ppm) in controlled environment conditions, and these were related to increases in below ground plant productivity acting as increased food resources (Newton et al., 1996; Yeates et al., 1997). An increase in predacious nematodes was also observed and it is possible this was mediated via an increase in their main prey, bacterial feeding nematodes (Yeates et al., 1997), which would have been responding to the increased root exudate resource via increased bacterial abundance (Ruf et al., 2006). The plant community composition was also affected in this experiment with an increase in the proportion of legumes present (Newton et al., 1996), and a subsequent increase in abundance of the obligate plant-feeding *Meloidogyne* nematodes (Yeates et al., 1997). This finding that appears to be common in eCO_2 conditions (Newton et al., 1995; Ross et al., 2004; Izaurralde et al., 2011) through a compensatory mechanism for progressive N limitation (Newton et al., 2010), especially where no additional nutrients are supplied.

In the grazed New Zealand pasture Free Air CO_2 Enrichment (FACE) experiment based in Bulls on Pukepuke black sand soil, soil faunal changes as a result of eCO_2 included a large increase in abundance of the obligate plant feeding nematode *Longidorus elongatus* (Yeates & Newton, 2009), which feeds on *Lolium perenne* as its preferred host (Boag & Geoghegan, 1984). The effect was observed both at 4 and 9 years after the eCO_2 treatment (475 ppm), suggesting this is a permanent shift in abundance in comparison to ambient CO_2 conditions. There was no concomitant change in yield of *Lolium perenne*, but it is possible that reduced *L. perenne* root biomass due to increased feeding by larger populations of the nematode (Yeates & Newton 2009) provided a negative feedback to plant yield under eCO_2.

Fungal feeding and omnivorous nematodes also showed consistent population changes between CO_2 treatments over time in the FACE experiment (Yeates & Newton 2009). Neither enchytraeid worms (Yeates & Newton, 2009) nor earthworms (Chevallier et al., 2006) showed significant populations changes in the long term, in contrast to earlier results of Yeates et al. (2003) that were a previous shorter term study from the FACE site. However, an analysis of earthworm casts showed they contained lower N concentrations under $eCO_{2,}$ apparently as a result of lowered N content of sheep dung (Chevallier et al., 2006). This suggesting that nutrients limitation in the plants may over time alter nutrient uptake of higher trophic levels.

In overseas FACE experiments, faunal changes have been observed with long-term eCO_2, including significant increases in abundance of ciliate protozoa, collembola and gamasida mites in sandy soil beneath ungrazed continental climate grassland in Minnesota (US), where CO_2 was increased by 180 ppm above ambient (Eisenhauer et al., 2012). While the changes in detritovore protozoa and collembola were likely due to changes in plant inputs, the gamasida mites are predominantly predatory with small soil forms predating on nematodes, so changes in mite abundance (ca 2× greater in eCO_2 than ambient) may have been in response to nematode population changes. However, the populations and taxa richness of nematodes were not observed to be significantly different between CO_2

treatments at this single sampling, so that temporal variation coupled with the small soil samples used for nematode determination (Eisenhauer et al., 2012) may have masked any effects.

Effects of eCO_2 on collembola have also been observed in a FACE experiment on cropping land in Germany (Sticht et al., 2006). In plots sown to wheat in a loamy sand soil where CO_2 was elevated to 548 (vs. 377 ppm for ambient), collembolan diversity and abundance increased, the latter by more than 50%. The ^{13}C results of Sticht et al. (2006) showed increased collembola abundance in the eCO_2 treatment that was most likely linked to increased plant root exudates via increases in rhizosphere fungal abundance. This is similar to the result for bacterial feeding nematodes observed by Yeates et al. (1997).

Ayres et al. (2008) showed that despite increases in plant root production, plant feeding nematode populations remained largely unchanged in eCO_2 treatments (ca. 700 ppm) applied to three grassland soils from Mediterranean-type climates (California and Colorado (USA), and Montpellier (France). There were increases in one family of plant feeding nematodes at one site and decreases in another at a separate site under eCO_2 treatments but, given the 20–30% increases in root biomass at the sites (Ayres et al., 2008), it is surprising more nematode population effects were not observed. Ayres et al., suggested that a lack of observable effect on herbivorous nematode populations despite increases in plant production might be due either to changes in plant root quality or to regulatory effects of predators and pathogens, and few studies have included assessments of these factors. Those studies where single population samplings have been taken, especially at the end of a crop growing season, may merely reflect the net result of changes in herbivore and subsequent predator and pathogen populations throughout the year. This approach would therefore overlook any temporal variation that might demonstrate plant-herbivore relations more clearly.

In a meta-analysis of largely unmanaged ecosystems, Blankinship et al. (2011) showed that eCO_2 had greater effects on the microfauna compared to other studied soil animals, leading to increases in abundance. They also showed that the effects of experimental eCO_2 across all soil biota lessened with increasing experimental duration.

Increases in temperature

Increased temperatures due to climate change are likely to have a range of effects on soil biota, depending on their biological thermal optima (e.g., Stamou et al., 1995; Verschoor et al., 2001; Stelzer, 2005; Šustr & Pižl, 2010). Increasing temperature could result in more generations per year for some soil invertebrates provided food and other resources were not limiting. This may have an impact on functioning of some parts of the soil community.

One of the most well-known impacts of soil warming on soil biota function is that of increasing populations and soil organic matter turnover by encyhtraeids in peatlands (Briones et al., 2007). However, this effect has often been observed in high latitude, cool climate peatlands where small changes in temperature have a disproportionately large impact on soil biota. The effect of warming will depend on the ecosystem in which it occurs, with cool, dry sites being more susceptible to change than other sites (Blankinship et al., 2011).

As well as potential impacts of soil warming on local soil biota function, global increases in temperature will potentially affect the distribution of soil organisms. Yeates and Boag (1998) used the climate matching function of the climate simulation model CLIMEX

(Sutherst & Maywald, 1985) to predict the spread of an herbivorous nematode (*Paratrichodorus minor*) within Australia and Africa. We have extended their analysis to New Zealand by using the matched location they employed (Grafton, NSW, Australia) and a more recent CLIMEX model (v 3.0). This was done after validating their findings with respect to Australia. *P. minor* is present in New Zealand and is known to have a localised distribution (Sturhan et al., 1997; Bell & Watson, 2001, pers. obs.). In the current analysis conducted for this report, we have used the fine resolution climate data for 1961–1990 available from the CliMond website (https://www.climond.org/Default.aspx; see Kriticos et al. (2012) for details of data derivation) along with the 2070 fine resolution data for the A1B and A2 climate change scenarios developed by CSIRO, Australia (CSIRO Mk 3.0) and the Centre for Climate Research, Japan (MIROC-H) respectively. These two combinations of scenarios and models were chosen to represent moderate and more severe climate changes respectively. The Combined Match Index level was set at 0.57 (with all but the relative humidity factor selected as contributing to the climate match) as this produced a current distribution most closely resembling that given by Sturhan and Yeates (1997) and observed from a number of national samplings (pers. obs.). Under the two climate change scenarios examined here, the *P. minor* distribution in New Zealand spread southward (Figure 3.14), with increasingly strong matches to those sites currently identified as being suitable for this nematode.

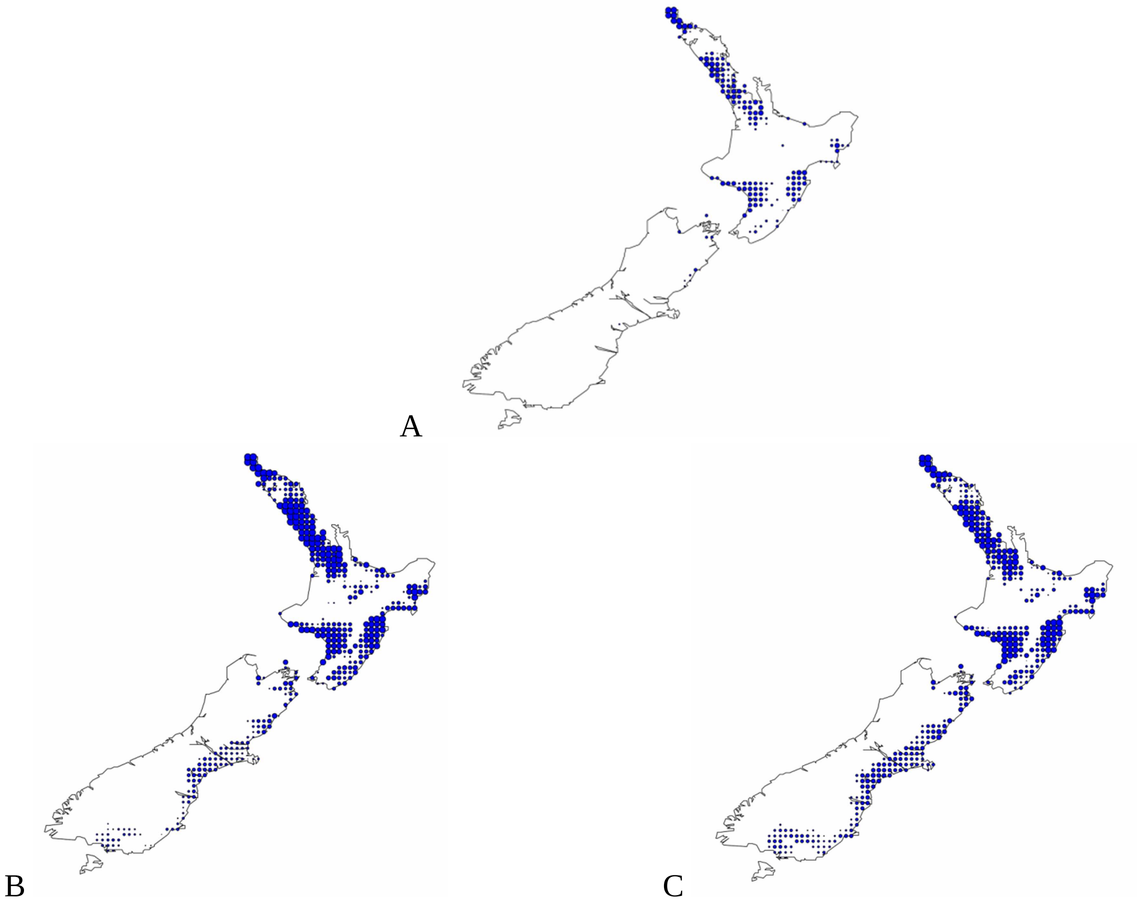

Figure 3.14: Distribution of *Paratichodorus minor* herbivorous nematode using Grafton, NSW as matching climate under: A) 1961–1990 temperature and moisture; B) Scenario A1B for 2070 using CSIRO model; C) Scenario A2 for 2070 using MIROC-H model.

P. minor feeding has only weak to moderate effects on plant yield, so its spread into other areas may not cause much extra loss of pasture production. However, the herbivorous nematode genera *Meloidogyne* is among the most damaging nematode pests globally, and in New Zealand there is at least one species with a currently restricted distribution which seem to be related to climate factors. *M. trifoliophila* is a clover-feeder that causes large reductions in yield of this important pasture component (Mercer et al., 2008) and is currently limited to sites in the North Island of New Zealand (Mercer et al., 1997; Bell et al., 2006). It has also been found in sites on the NSW and Queensland border in Australia (Zahid et al., 2001), and in the south western corner of Tennessee in the US (Bernard & Jennings, 1997). By paramatersing the species model within CLIMEX for *M. trifoliophila* using data published for a similar species found in New Zealand (Mercer & Grant, 1993) and adjusting abiotic stress variables in the model it was possible to calculate the CLIMEX Ecoclimatic Index (EI; see Svobodová et al., 2013) for sites in Australia and the US (Figure 3.15). The EI gives an indication of suitability of sites for establishment and long-term persistence of an organism and, in this case, gave distributions in Australia and the US in line with observed distributions.

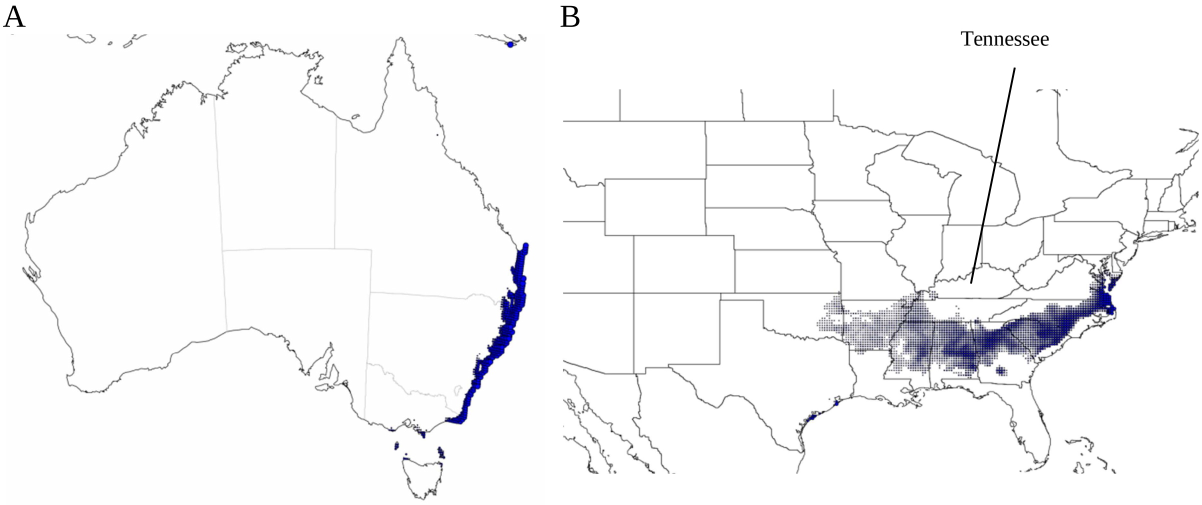

Figure 3.15: Distribution of *Meloidogyne trifoliophila* herbivorous nematodes based on CLIMEX Ecoclimatic Index>0: A) Australia; and B) USA using 1961–1990 temperature and moisture.

Extending this analysis to New Zealand (Fig. 3.16) shows that the calculated site EIs with no climate change scenario added to the model gives good agreement with the observed current New Zealand *M. trifoliophila* distribution, which is from Northland to Palmerston North and Wellington (Mercer et al., 1997). The climate change A1B scenario (data for which includes changes in both temperature and rainfall) shows *M. trifoliophila* EI scores becoming stronger for the North Island’s central plateau and an extended distribution into the Wairarapa. In the South Island the calculations show areas suitable for *M. trifoliophila* establishment and survival spreading down both east and west coasts as far south as Canterbury and Greymouth. The A2 scenario shows limited extension of possible range from the A1B scenario, but the increasingly large EI scores indicate increased likelihood of establishment and survival in the areas indicated by the A1B scenario.

Nematodes have limited self-dispersal capabilities, so human-assisted dispersal would be the main route for extensions in distribution to areas that become suitable. Dispersal of soil would be the highest risk pathway for human-assisted movement of *Meloidogyne* nematodes, so any measures that limit soil movement, particularly between North and South Islands, would help reduce spread of these pests.

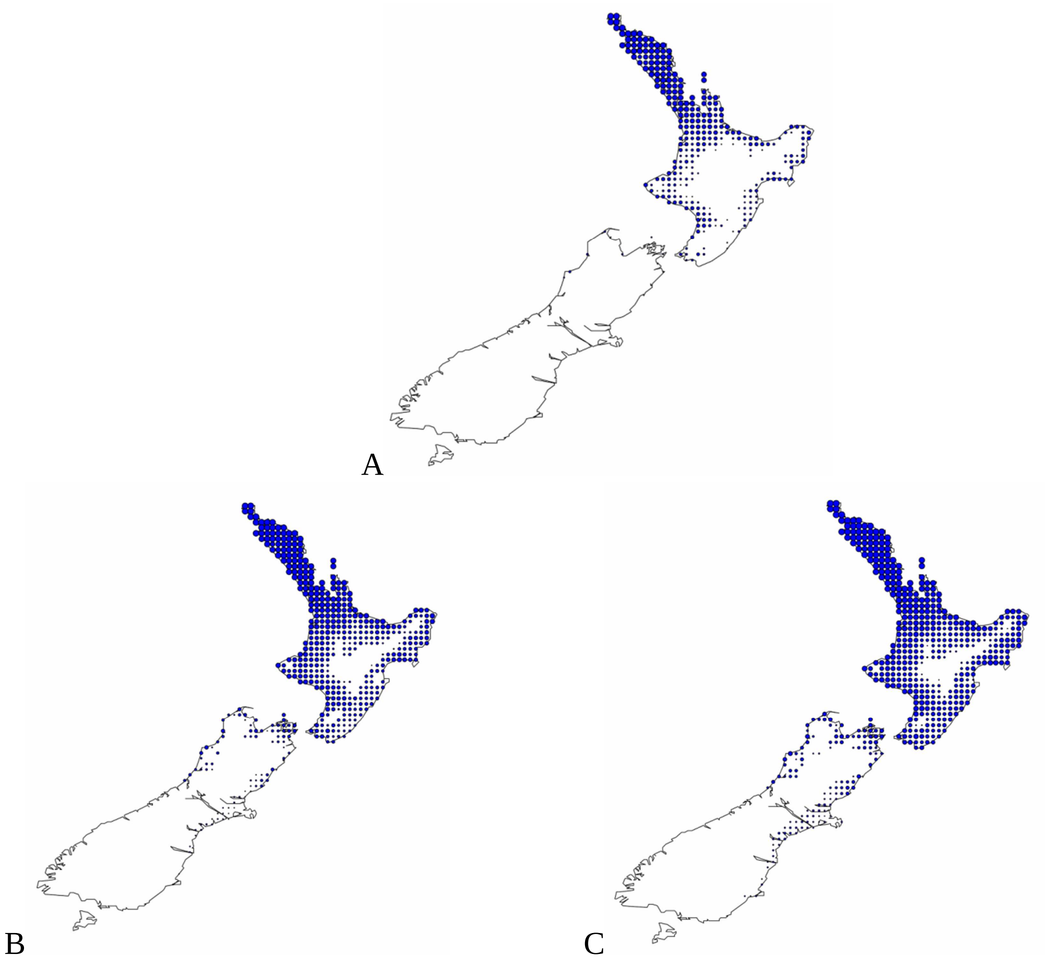

Figure 3.16: Distribution of the herbivorous nematode *Meloidogyne trifoliophila* based on CLIMEX Ecoclimatic Index>0: A) 1961–1990 temperature and moisture; B) Scenario A1B for 2070 using CSIRO model; C) Scenario A2 for 2070 using MIROC-H model.

Changes in moisture availability

The impact on soil biota of changes in level and distribution of precipitation has been investigated either alone or as interactions with other climate change factors such as CO_2. For example, Kardol et al. (2010) used a Tennessee long-term climate change experiment with old-growth sub-shrub and *Fescue* plants, to study impacts of increased moisture, temperature and CO_2 on soil nematode communities. They found that increased moisture had a greater effect than increased temperature or CO_2. Increased moisture resulted in increased populations of total nematodes and all feeding groups (except fungal feeders) in soil from random between-plant spaces, and most feeding groups (except ectoparasites) in soil beneath specific plants. However, the moisture level was ca. 12 times higher in the high treatment than the low, which equated to about 10% greater soil moisture by volume, and it is not clear if this increase is realistic under current climate change scenarios. Increased moisture also resulted in increased plant biomass, so it is possible the observed nematode effects were mediated via plant productivity and litter inputs. There was a significant positive CO_2 effect only on nematode root hair feeders in the between-plant soil, and a significant negative temperature effect only on bacterial feeders in the between-plant soil and endoparasites in plant-specific soil. This study calculated abundance rather than percent composition; the percentage data may have helped clarify whether the changes observed were changes in nematode community composition or merely scaling effects from an increased total nematode abundance.

Reductions in mite and collembola abundance have been observed with reduced precipitation in Minnesota grassland (Eisenhauer et al., 2012). Although enchytraeid populations did not decline with increased drought intensity and frequency in heathland sites in Wales, the Netherlands, and Denmark, there was a strong and persistent effect on species richness and community structure, which was strongest at the driest sites (Holmstrup et al., 2012). Negative impacts of decreased moisture and increased temperature regimes on nematode and protozoa total populations have been observed in Danish heathland (Stevnbak et al., 2012). Similarly, enchytraeid populations were reduced with decreased moisture. However, the soil samples used for nematode determinations in that study were rather small (3–16 g), making it more difficult to generalise results. In a low rainfall cool temperate fescue grassland in Hungary, the largest effect on nematode abundance was soil moisture, with increased temperature impacting on community diversity (Bakonyi et al., 2007).

From their meta-analysis, Blankinship et al. (2011) concluded that changes in precipitation would have positive impacts on soil biota abundance in forest ecosystems and for the mesofauna. They also showed that the effects of altered precipitation significantly decreased with increasing mean annual temperature, presumably through an effect of temperature on evapotranspiration. The effects of changes in precipitation also increased with the duration of experimental studies, suggesting that community, and presumably function, effects are permanent.

Indirect effects

Elevated CO_2 has differential effects on earthworm (*Lumbricus terrestris*) biomass depending on the the diversity of the plant community in defaunated grassland soil microcosms maintained in glasshouses in the UK (Milcu et al., 2011). For microcosms with the most diverse plant communities (8 vs. 4 or 1 species), earthworms lost significantly less biomass under eCO_2 (600 ppm) conditions compared to ambient CO_2. It was suggested this was due to the significantly greater soil moisture in the eCO_2 treatments, coupled with greater soil N in more diverse plant community microcosms (Milcu et al., 2011).

Of course, not only do climate change factors interact to impact on the soil biota, but different components of the soil biota themselves interact, and those interactions may also be modified by climate change. Uvarov (2009) showed in a laboratory experiment that the impact of earthworm feeding on the detrital nematode community changed with variations in diurnal temperature fluctuations. Earthworms also have impacts on microbial C breakdown and the magnitude of these impacts vary with temperature in a non-linear fashion (Wolters & Ekschmitt, 1995).

It is also clear that not only do climate change factors impact on soil biota but that the reverse also occurs: soil biota have an influence on climate change factors. For example, earthworms are known to emit N_2O and N_2 as part of their digestion process (Karsten & Drake, 1997; Wüst et al., 2009), but it is likely the impacts are small.

Climate change scenarios predict an increase in extreme weather events in some areas resulting, for instance, in the increased frequency and severity of flooding and droughts. It has been suggested that earthworms would be impacted by decreased intervals between flooding via a reduction in their ability to advance from cocoons through to reproductive adults, possibly leading to local extinction (Thonon & Klok, 2007). In areas where flooding is currently a problem, a drying trend from climate change would make these areas more amenable to earthworm survival.

The indirect effects of alteration of soil biotic communities by climate change are further discussed in section 3.6.

Microbial communities

Although soil microbial communities regulate many functions associated with soil nutrient cycling and plant health (Figure 3.17), there is no straightforward answer to the question of how provision of soil ecosystem services will be impacted by climate change. Understanding how climate change will affect soil community composition and biodiversity and resultant effects on functions such as nutrient cycling will be needed before it will be possible to predict the fate of global C and N cycles (Pritchard, 2011) and functioning of New Zealand's natural and managed ecosystems. Soil ecosystems are comprised of complex communities of bacteria, archaea and eukaryotic taxa. The diversity (species richness) of soils is incredibly high; in fact the highest of any ecosystem known (Curtis et al., 2002). Estimates of the diversity of life in soil vary widely but for the bacteria alone a general consensus of richness is in the order of thousands to tens-of-thousands of species per gram of agricultural soil (Curtis et al., 2002; Torsvik et al., 2002; Schloss & Handelsman, 2006).

Microbial communities play key roles in determining rates of important ecosystem processes such as trace gas formation (Schimel & Gulledge, 1998), C cycling and sequestration (Waldrop et al., 2000; Schimel & Schaeffer, 2012), decomposition (Hendrix et al., 1986), soil N biogeochemistry (Balser & Firestone, 2005; Cavigelli & Robertson, 2000), and disease suppression (Mendes et al., 2011). Many of the microbial species present in soil are at low abundance, i.e. are considered to be part of the ‘rare biosphere’ (Elshahed et al., 2008). However, even these numerically rare taxa have been shown to be highly responsive to changes in soil conditions, and to play an important role in supporting ecosystem processes, such as the biogeochemical cycling of sulphur (S; e.g. Pester et al., 2010) and N (e.g. Mertens et al., 2009).

While numerous studies have sought to understand projected impacts of climate change on soil biology and function, many have been conducted in systems such as tropical forests, tundra, deserts and so forth, far fewer have investigated impacts in temperate agricultural and grassland ecosystems that are relevant to agroecological conditions prevalent in New Zealand. Here we assess the potential impacts of climate change with a focus on temperate soils under grassland and agricultural use.

Soil type is a key driver affecting the structure and diversity of most soil microbial communities. This is, for a significant part, driven by pH-related effects that control the range of bacteria and fungi present. However, within the existing range of soil types present in New Zealand, and considering adjustment of soil pH as a key aspect of farming system management, the effects of climate-associated drivers are likely to be important: changes in temperature, precipitation, alteration in CO_2, and increased variation in rainfall, temperature, and occurrence of extreme events. Also important are secondary effects: increased intensification of farming systems, alteration of land use, use of different crop and pasture species, adoption of new farming management practices, and so forth. Because it is difficult to speculate on the nature and extent of these secondary effects, we focus on potential impacts of factors directly associated with climate change.

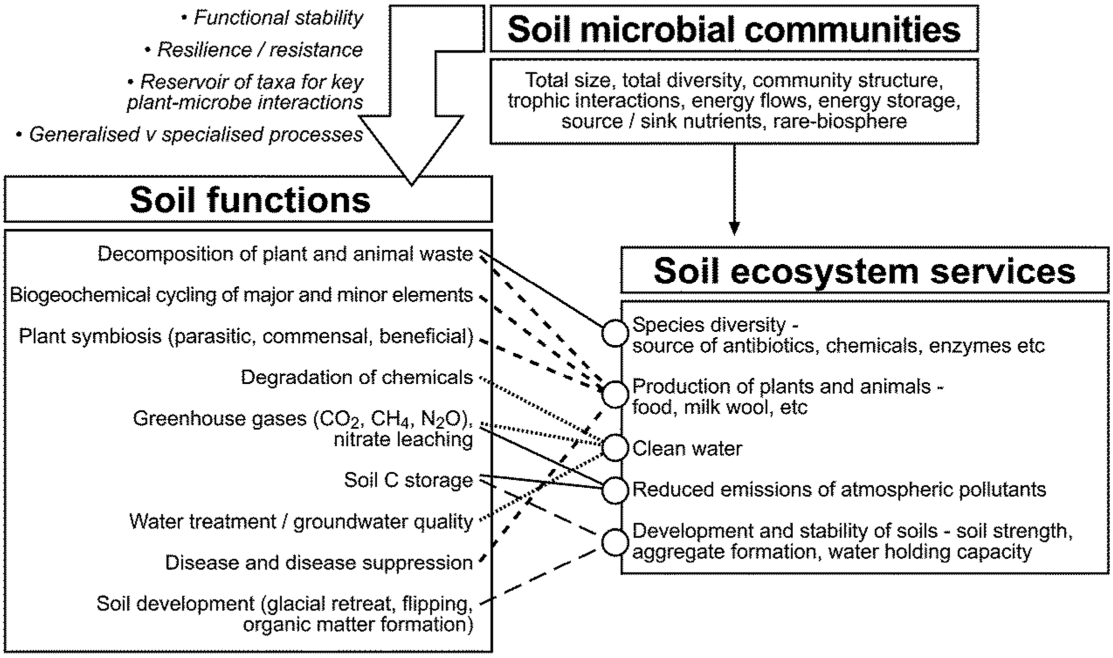

Figure 3.17: Conceptual diagram of how soil microbial communities and the diversity of species and community composition therein support soil functions and thereby soil ecosystems services.

Methodology affects study outcomes

Methods used to assess impacts of climate change on microbial community structure and function vary widely, and the level of resolution/sensitivity of methods can impact conclusions drawn.

In an Australian native grassland soil, the impacts of elevated temperature (ambient plus 2°C) and CO_2 (ambient plus 550 ppm) on soil bacterial, archaeal, and fungal communities was examined (Hayden et al., 2012). The study was conducted at a long-term climate change experimental site (TasFACE). It attempted to link the microbial community to variation in soil nutrient status and physicochemical conditions occurring as a result of altered plant growth using a combination of low- and high-resolution molecular methods. Findings associated with the archaeal community composition varied with analytical method; the relatively low resolution approach of terminal restriction fragment length polymorphism (TRFLP community fingerprinting) showed strong separation of community structure by CO_2 enrichment, yet no separation was observed when the PhyloChip microarray tool was used. Fungal community composition, assessed only with TRFLP, was responsive to CO_2, a combination of warming x CO_2, and a range of other variables. Bacterial community structure, when assessed using TRFLP, was not influenced by CO_2 or warming, but when assessed using the high-resolution microarray approach, specific influences on various phyla were found. At the phyla level, Actinobacteria and Alphaproteobacteria increased with warming, while Bacteriodetes and Firmicutes increased with CO_2 enrichment. A number of operational taxonomic units (OTUs) varied in response to CO_2 x temperature. These findings demonstrate that taxonomic resolution is important when investigating the impacts of second-order influences on microbial community assemblage. In particular, high-resolution tools may be required to unravel subtle changes in community composition, particularly in low nutrient / low input systems such as native grasslands.

Microbial community response to increased temperature and atmospheric CO_2

In an experimental study by Bardgett et al. (1999) the microbial community response to change in ambient temperature of 2°C was measured in a low input soil system (Ecotron). The experiment was run over successive plant generations to allow for the ongoing succession development of the microbial community with regards to the plant life cycle. In the first generation, increases in fast-growing bacterial species (‘r’ strategists) were detected under the high-temperature regime, but other taxa, including fungi, were unchanged. Plant inputs into the soil were also unchanged, indicating that the bacterial response detected was directly temperature related. Overall, in the low nutrient status soils, the effect of temperature on the microbial community was considered negligible. Similar findings were found in an Ecotron study by Kandeler et al. (2002), where the influence of temperature (ambient plus 2°C) and CO_2 (ambient plus 200 ppm) on soil microbial communities was assessed. In a field-based study, Niklaus et al. (2003) undertook comprehensive assessment of biology in soil undergoing CO_2 enrichment (ambient and 1.7x ambient) over six growing seasons. Microbial biomass, community composition, nematodes, and microarthropods were assessed alongside soil physicochemical properties. The community of soil microorganisms did not respond to CO_2, either in terms of biomass, composition, or metabolic quotient (qCO_2).

Interactions are important yet complex

The magnitude of effects of climate change associated drivers on soil microbiology is highly interactive, and the interactions can be differentially expressed on the various components of the soil ecosystem.

The Old-Field Community, Climate, and Atmosphere Manipulation (OCCAM) experiment has investigated the above and belowground impacts of CO_2 enrichment, rainfall exclusion, elevated temperature (+3°C over ambient), and irrigation (Castro et al., 2010; Gray et al., 2011). They found that (i) fungal abundance increased with temperature, (ii) bacterial abundance increased in warm plots but only when CO_2 was elevated, and (iii) different taxa within the bacterial and fungal populations varied (+/-) with precipitation. In a scenario with increased temperatures, increased CO_2 and increased precipitation, there would therefore be an overall shift towards a bacterial dominated system, but the types of bacteria and fungi present would also vary within this.

Cheng et al. (2011) investigated the impacts of eCO_2 in soils undergoing N addition. Under conditions of eCO_2 (1.5x ambient), increased plant residue C and N inputs were deposited to the soil and, concomitantly, microbial biomass and heterotrophic respiration were elevated. These effects manifested after several years of the trial commencement, emerging as the soil N status increased. Thus, compared with studies conducted in low fertility soil ecosystems, the impacts of CO_2 enrichment in improved (fertilised) agroecosystems may be significantly enhanced.

Hu et al. (2006) conducted an analysis of experimental studies investigating the role of N on plant residue inputs and microbial decomposition. The findings showed that the initial ecosystem N status was an important attribute regulating N biogeochemistry under eCO_2 conditions. The effects of CO_2 in improved agricultural systems will, therefore, differ to those in non-improved systems.

Effect of precipitation patterns

Climate change models predict an increase in the variation of rainfall patterns, leading to increased variation in soil moisture content. The impacts of drying-rewetting frequency are potentially an important driver of alteration in microbial communities.

Fierer et al. (2003) examined the impacts of drying-rewetting on community structure in soil under grassland and found no evidence for change in the community structure with increased frequency of stress. However, effects on bacterial communities were found in soils under oak canopies when exposed to wet-dry stress (Fierer et al., 2003). Bapiri et al. (2010) explored dry-rewetting impacts on bacterial and fungal communities. Fungi were generally more tolerant of the impacts of drying than the bacteria, resulting in an increase in fungal dominance. The findings demonstrate that the impacts of drying-rewetting vary between the evaluated systems and are likely to be due to factors associated with the environmental history of the samples (i.e. if they are normally not exposed to such impacts), and/or the original composition of the microbial community. This is reflected in studies that find that land-use management, whether extensively or intensively managed, changes the soil microbial community structure and functional composition (Wakelin et al., 2013), and impacts soil food web structure and the resistance and resilience of trophic communities to climate change (de Vries et al., 2012).

The potential impacts of warming appear to be strongly influenced by precipitation patterns. Shiek et al. (2011) conducted a study on the soil bacterial community at a long-term global warming experimental field site (KFFL; Kessler Farm Field Laboratory). The trial site has replicated plots with treatments artificially held at ambient and ambient + 2°C. In years receiving normal precipitation, warming increased the soil microbial population size overall (40–150%), but significantly reduced the richness and evenness of bacterial species and also altered community composition. However, in years of low rainfall, the elevated temperatures resulted in strong moisture deficit and reduced plant production; this combination of factors also impacted the soil bacterial community. This study is important as it highlights the significance of rainfall in the regulation of the microbial community response to elevated ambient temperatures. Furthermore, as the biological components of the ecosystem differed significantly in response to to the climate-related forcings, this is likely to have wide and disruptive impact on overall soil function.

Interactions between soil moisture and temperature and eCO$_2$

Interactions between soil moisture content and temperature, or soil moisture and CO$_2$, have clear impacts on the soil microbial community (Shiek et al., 2011). Under elevated temperatures and in dry soils, the abundance of Firmicute-type bacteria increased; these bacteria are stress tolerant and can form resistant structures. The less hardy gram-negative bacteria decreased in abundance in the dry samples. Under elevated temperatures the abundance of saprophytic fungi was decreased, which may have been indirectly driven through increases in nematode abundance. The importance of trophic interactions on down- or up-regulating the impacts of temperature on saprophytic fungi has also been demonstrated in other soil ecosystems (A’Bear et al., 2012). Such indirect effects, evident here though food web interactions, add to the level of overall complexity in understanding climate-change impacts on soil microbiology. Certainly ecosystem-level approaches are required.

Various groups of soil taxa respond quite differently to climate-associated influences. Here we examine the climate change effects on two specific groups of organisms that are important for the functioning of pastoral ecosystems. Effects on specific taxa with close root association may have important outcomes for plant health and ecosystem productivity, but may be non-observable when assayed using tools at high taxonomic level (e.g., phylum level investigations).

Microbial symbionts - Rhizobia

There is strong evidence that increases in atmospheric CO_2 will lead to increases in biological N_2 fixation (BNF) in the legume-rhizobial symbiosis in agricultural systems. Typically, more N is fixed because of an increase in plant size, resulting in greater nodule mass per plant, rather than an increase in N fixation per unit nodule mass (Edwards et al., 2006).

A recent meta-analysis of N dynamics in grain crops and legume pasture systems under elevated atmospheric CO_2 concentrations reported a 38% increase in the amount of N fixed by legumes, which was accompanied by increased numbers of nodules (33%), nodule mass (39%), and nitrogenase activity (37%; Lam et al., 2012). Under levels of increased CO_2, plant photosynthesis increases resources to legume nodules that can result in increased N-fixation by the rhizobial symbiont. Such effects have been observed on lucerne under eCO_2 conditions in the Swiss FACE trial, where effectively nodulated plants increased by 50% compared with plants in the ambient control (Luscher et al., 2000). Similarly, increases in populations of N_2-fixing bacteria rhizobia in the rhizosphere of white clover plants exposed to eCO_2 have been reported (Schortemeyer et al., 1996) and increases in the size and number of root nodules on soybean (Serraj et al., 1998) have been reported in soils exposed to increased atmospheric CO_2. However, these effects may not be universal, but rather vary with legume species and N status of the soil (West et al., 2005). In natural (unfertilised) soil ecosystems, the ability of legumes to respond to eCO_2 can be limited by supply of other nutrients, such as P, thereby reducing the expression of this 'CO_2 fertilisation' effect (Rogers et al., 2009). Furthermore, there is a risk that as atmospheric CO_2 levels increase, enhancement in legume biomass through increased symbiotic N-fixation will only occur when the host-symbiont relationship is efficient at N_2-fixation; if they are poor N-fixers, negative growth responses may occur and this may be influenced by soil N content (Luscher et al., 2000).

New Zealand pastoral agriculture is highly dependent on the N-fixation through the symbiosis between white clover *Trifolium repens and* Rhizobium (Ledgard, 2001; Parfitt, 2006). BNF by white clover has been found to increase under eCO_2 in controlled environments (Ryle et al., 1992), open top chambers (Manderscheid et al., 1997), and in the field (Zanetti et al., 1996), under non-limiting soil nutrient conditions. In contrast, in the only relevant study carried out in New Zealand, Watanabe et al. (unpublished) found lower numbers of *nifH* genes and gene transcripts in nodules on white clover growing under eCO_2 in the FACE experiment (Newton et al., 1996), which corresponded to reduced proportion of atmospheric N fixed under these conditions. There was no difference in nodule numbers, and the strains present were the same in both treatments. The mechanism of inhibition is not known, but there is evidence that P is as limiting as N at the NZFACE site (Gentile et al., 2012). Edwards et al. (2006) found that white clover grown in a mixed sward with buffalo grass under eCO_2 fixed significantly more N than plants grown under ambient CO_2 but only in the presence of high soil phosphorus. The authors suggested that where soil P availability was low, eCO_2 would not increase BNF and pasture quality could decrease because of a reduction in aboveground N. Given New Zealand's reliance on the white clover/Rhizobium symbiosis, these findings warrant further investigation.

Ammonia oxidising bacteria

Ammonia oxidising bacteria (AOB) convert NH_4^+ to NO_3^-. Because this is the rate limiting step in nitrification, AOB have a central role in global N cycling. AOB are an ideal model group for microbial ecology studies as they are ubiquitous, of significant environmental importance as a functional group, sensitive to environmental changes including simulated global changes, and carry the gene *amoA*, which can be used as a molecular marker, allowing detection and quantification of AOB without direct need for laboratory culture (Kowalchuk & Stephen, 2001; Niklaus & Kandeler, 2001).

Soil AOB community composition was assessed in soils collected from long term concentration gradients established by CO_2-emitting vents from natural cold CO_2 springs in Northland, New Zealand (Bowatte et al., 2008). The site had been exposed to eCO_2 for several decades. Soil nitrification activity decreased with increasing CO_2 levels, in association with a change in the composition of the AOB community, with the response probably due to changes in the input of soil C from the plants. Increased flux of C to soil at eCO_2 has been shown to stimulate growth of heterotrophs, which can out-compete AOB for available NH_4^+.

Horz et al. (2004) previously examined the effects of multiple climate change parameters (increased atmospheric CO_2, precipitation, temperature, and N deposition) on the AOB community in grassland soil. Their study reported changes in AOB community structure and a decrease in total abundance of AOB in grassland soil exposed to eCO_2, thought to be related to competition with heterotrophic bacteria. The decrease in AOB abundance was most pronounced when precipitation was increased. Increases in nitrification were associated with shifts in the AOB community but not changes in abundance.

A more recent study (Malchair et al., 2010) examined the effects of plant species richness and a 3°C increase in ambient air temperature on potential nitrification, basal respiration and AOB communities in an experimental grassland scenario. Plant species mixtures were derived from nine species of three functional plant groups: forbs, legumes, and grasses. AOB richness and community structure were not affected by warming but were associated with plant species richness, indicting the important role of plants species in structuring the soil microbial community. There was no clear relationship between AOB richness and potential nitrification. Avrahami et al. (2003) had previously reported that temperature was a selective factor for AOB, but that study looked at larger temperature shifts (>5^{o}C), unlikely in current mid-century climate change scenarios forecast for New Zealand.

Rhizosphere bacteria and Pseudomonas

Microorganisms in the rhizosphere appear to be more impacted by CO_2 than those in the bulk soil. Using samples taken from under pasture in a FACE experiment, Montealegre et al. (2002) demonstrated that bulk soil communities were largely unaffected by CO_2. However, metabolically active bacteria increased by 170% in the rhizophere of CO_2-treated plants, presumably due to either increases in root exudation or more rapid turnover of the plant roots.

Pseudomonas bacteria are important root colonisers with widely differing phenotypes and functions, ranging from beneficial behaviour such plant growth promotion and suppression of plant disease, through to causing a range of plant diseases.

Pseudomonas populations were shown to increase in frequency in the rhizosphere of perennial ryegrass after 3 years under elevated eCO_2, but there was a decrease in the rhizosphere of white clover (Marilley et al., 1999). Frequency of selected functions carried out by *Pseudomonas* strains (production of auxins, siderophores and hydrogen cyanide – indicative of biocontrol strains) were tracked in *Pseudomonas* populations associated with two perennial grasses (*Lolium perenne* and *Molinia coerulea*) grown at ambient and eCO_2 in the Swiss FACE experiment (Tarnawski et al., 2006). Frequencies of siderophore producers and nitrate dissimilating strains were higher and those of hydrogen cyanide producers lower under eCO_2, demonstrating the potential for selection of particular bacteria that may impact on the fitness of the plant, and thereby affect primary productivity, in the longer term.

Plant pathogens

Energy flow through fungal pathways may be enhanced relative to bacterial pathways in response to both warming and atmospheric CO_2 enrichment. Whether fungal domination of soils will lead to increased severity of soil borne fungal plant diseases is unknown (Pritchard, 2011). Soil-borne plant pathogens account for significant crop losses and can often be more difficult to control than foliar diseases, but it is currently unclear how climate change will affect plant-pathogen interactions. Outcomes are likely to be different depending on the plant-pathogen combination. The geographical range of some plant pathogens depends on environmental variables, most often temperature (Shaw & Osborne, 2011).

Microbial community summary

Soil microbial communities drive processes and functions that deliver soil ecosystem outcomes (Fig. 3.1). The rich diversity of microbial species in soils is essential as it supports the capacity of the ecosystem to change with varying selective pressures (Figure 3.17), and provides necessary layers of niche overlap and functional redundancy that facilitate ecosystem resilience and resistance. Shifts in the diversity or structural composition of communities, whether by climate change or other factors, may therefore impact on the provision of soil ecosystem services.

Based on the ecology and biogeography of soil microbial communities, it is clear that climate-related factors, for example temperature range and soil moisture deficit, are important controls affecting microbial life in soils. As such, variation in climatic conditions is likely to influence soil microorganisms, and impact soil function. This has been experimentally validated in numerous studies (laboratory microcosms and field-based FACE studies). Furthermore, the findings of these studies project that the strongest effects are likely to be seen in improved (fertilised) soils; this means that microbial life and processes in agricultural soils are particularly susceptible to climate change influences. Following this, the movement from extensive land use, such as grazing of grasslands, to intensive agricultural production is likely to result in soil ecosystems more susceptible to climate change impacts.

Fungal Communities

Soil fungal communities divide into three major functional groups based on their source of carbon: mycorrhizal fungi, saprotrophic fungi, and pathogenic fungi.

Mycorrhizal fungi

Most agricultural and horticultural species live mutualistically with arbuscular mycorrhizal fungi (AMF), including rye grass, most grains, clover, and most horticultural crops. The fungi can represent 5–50% of the microbial biomass in soil underneath these crops, with the wide range reflecting uncertainty of measurement (Olsson et al., 1999). Regardless of the exact quantity, supporting mycorrhizal biomass comes at a C cost to the host plant on the order of 5–10% of total photosynthetic C gain (Bryla & Eissenstat, 2005; Fitter et al., 2011). In many crops the biomass of arbuscular mycorrhizal fungi is reduced by high P levels from fertilisation, and no plant growth enhancement by arbuscular mycorrhizas is observed (Ryan & Graham, 2002). Nonetheless, up to 100% of the plant P uptake may be obtained through mycorrhizal hyphae even where no plant growth enhancement by arbuscular mycorrhizal fungi is observed. As such, mycorrhizal fungi are probably best understood as a part of the soil–plant ecosystem, regardless of positive or negative effects on plant growth.

Mycorrhizas are also very important in the forestry sector: Pines, Douglas-Fir, Eucalyptus, and Beech all associate with ectomycorrhizal fungi. All these trees are highly dependent on ectomycorrhiza (EcM), as seen in the failure of many early plantings of non-native tree species without their co-evolved mycorrhizal fungi (Dickie et al., 2010). At present ectomycorrhizal fungal inoculum is widespread in commercial production nurseries, hence a lack of mycorrhizal infection no longer is a limitation on tree establishment by planting (Walbert et al., 2010).

Sensitivity of mycorrhizal fungi to elevated C

Although it is often expected that mutualisms will be negatively affected by global climate change, this finding is based largely on pollination and seed dispersal mutualisms, and may not apply to mycorrhizal mutualisms (Tylianakis et al., 2008). Several studies show increased AMF and EcM biomass with eCO_2 (Lukac et al., 2003; Hu et al., 2005), which appears to be a fairly common, but by no means certain, outcome (Tylianakis et al., 2008). As mycorrhizal fungi link directly to plants, it can be difficult to separate indirect effect of CO_2 via enhanced plant growth from any direct effects of CO_2 on mycorrhizal fungi (Fitter et al., 2000). Free-air CO_2 enrichment experiments in Minnesota, USA, suggest indirect effects of eCO_2 on AMF (Antoninka et al., 2011). Interestingly, the effect on AMF biomass was primarily a CO_2 effect on plant growth regardless of species, while the effect on community composition of AMF was largely via the composition of the plant community.

Increased mycorrhizal fungi can enhance plant nutrient acquisition through increased competitiveness with soil microbes (Hu et al., 2005). Any resulting increase in above-ground productivity and C may be constrained by soil nutrient levels or other limiting resources, such that an initial increase in productivity is not maintained. The effects of increased mycorrhizal biomass on soil C may also be time-scale dependent. In the short term, an increased AMF biomass may result in net soil C loss, not gain (Cheng et al., 2012; Kowalchuk, 2012). However, models suggest that longer term accumulation of recalcitrant compounds derived from mycorrhizal fungi may result in net C gain (Verbruggen et al., 2012)

Sensitivity of mycorrhizal fungi to increased temperatures

As with elevated C, it is difficult to separate direct effects of temperature on mycorrhizal fungi from indirect effects via plants. Controlled environment growth chamber experiments suggest that increased temperature can increase AMF abundance and root infection, but not all fungal species are responsive to temperature (Staddon et al., 2004). Similarly, in EcM it has been shown that different fungal species vary substantially in their ability to acclimate to higher temperatures (Malcolm et al., 2008).

A key point in understanding the variability in mycorrhizal responses to elevated temperature is that mycorrhizal fungal communities are highly diverse, with as many as 100 species of fungi for every one species of associated plant (Dickie, 2007). This diversity may be key to the resilience of fungal functioning in the face of global climate change. Productive forests in New Zealand are atypical in having relatively low fungal diversity (Walbert et al., 2010), reflecting the non-native origin of pine and Douglas-fir and their limited capability to utilize native fungi (Dickie et al., 2010). This may make the New Zealand forestry sector more vulnerable than might be inferred from studies of pine or Douglas-fir responses in the Northern Hemisphere.

Sensitivity of mycorrhizal fungi to altered rainfall and extreme events

As with temperature, the response of mycorrhizal fungi to altered rainfall tends to vary across fungal species (Staddon et al., 2004). Several literature reviews suggest that mycorrhizal fungi tend to make plants more resistant to drought, independent of mycorrhizal effects on plant nutrient uptake (Entry et al., 2002; Ruiz-Lozano et al., 2012). Although the mechanism for greater drought resistance is uncertain, it appears that the mycorrhial association physiologically alters the plant so that it is less susceptible to drought stress.

All fungi produce hydrophobic proteins in their hyphae, and many fungi also have hydrophobic exudates (Rillig, 2005; Spohn & Rillig, 2012). Hydrophobicity of soil can be induced by drying and can subsequently reduce the ability of a soil to re-wet, increasing the risk of surface flow and runoff. As droughts are likely to increase with global climate change, better understanding of the links between fungi, hydrophobic proteins, and soil water repellency should be a high priority for future research.

Saprotrophic fungi

Traditionally, it has been assumed that decomposition in agricultural soils was dominated by bacteria, with very limited role of fungi. While some studies support this view (e.g., Macdonald et al., 2009), other studies have found that 50% or more of soil microbial biomass or activity is fungal, with particularly high fungal dominance in no-tillage crops (Frey et al., 1999; Wardle et al., 1999). Molecular methods have also revealed a high diversity of saprotrophic fungi in agricultural soils. For example, a sampling under maize crops from a single location in North America found nearly 100 species of basiodiomycete fungi (Lynch & Thorn, 2006). A similar study in Europe found 115 species of fungi across five agricultural soils, but no more than 34 species in any one soil (Klaubauf et al., 2010). The difference may reflect substrate quality, with maize producing a more woody detritus than other crops. High fungal:bacterial ratio soils tend to have lower N leaching and lower NO_2 emissions than low fungal:bacterial soils. In addition to recycling of soil nutrients, saprotrophic fungi may play a role in suppression of pathogenic fungi (Sarathchandra et al., 2005).

Sensitivity of saprotrophic fungi to eCO_2

There are unlikely to be any direct effects of eCO_2 on saprotrophic fungi, as these fungi typically occur in microhabitats with extremely high CO_2 levels.

Sensitivity of saprotrophic fungi to increased temperature

It is probably safest to state that we have no idea how saprotrophic fungi will respond to increased temperature. Experimental warming has shown large increases (Zhang et al., 2005; A'Bear et al., 2012), large decreases (Frey et al., 2008), or no change in the fungal:bacterial ratio (Bergner et al., 2004; Schindlbacher et al., 2011). Where increases in fungal dominance are observed, they can be sensitive to herbivory (Zhang et al., 2005) or fungivory (A'Bear et al., 2012).

Sensitivity of saprotrophic fungi to altered rainfall and extreme events

There are few systematic studies of fungal responses to soil moisture (Rousk & Baath, 2011). Nonetheless, at least one study found that soil fungi are more responsive to moisture than soil bacteria, resulting in increased biomass and greatly increased relative dominance of fungi with increased soil moisture (Frey et al., 1999). As with mycorrhizal fungi, increased drought may cause the development of soil water repellency, at least partially due to hydrophobic proteins of fungal origin.

Pathogenic fungi

Fungi and fungi-like soil organisms (e.g., *Phytophthora*) are very common causes of plant disease, with significant costs for the productive sector in New Zealand. About 30% of emerging infectious diseases are fungal. New Zealand is particularly sensitive to pathogens due to reliance on low diversity of plant species (e.g., kiwi fruit, *P. radiata*), making catastrophic economic loss through a single emerging pathogen possible. Between 1988 and 1997, three new fungi were recorded in the Auckland area per year (Ridley et al., 2000). There are predictions that global climate change will contribute not just to the spread and effects of existing disease, but also to the rate of emergence of new diseases (Anderson et al., 2004). The recent example of Psa disease in kiwi highlights the vulnerability of some sector of the economy. The disease was first detected in Nov 2010, and by Jan 2013 more than 6500 kiwi vines across more than 2000 orchards had been affected (http://www.kvh.org.nz), with an expected cost of $310–$410 million over the next five years (Greer & Saunders, 2012).

Sensitivity of pathogenic fungi to eCO_2

Increased CO_2 can modify plant foliar physiology in ways that increase resistance to pathogens (Chakraborty et al., 2000). Despite this, an increase in plant pathogens is often observed with increased CO_2 (Tylianakis et al., 2008)

Sensitivity of pathogenic fungi to increased temperatures

Overwintering soil temperature is a key limitation on many pathogenic species (Pfender & Vollmer, 1999; Garrett et al., 2006). Milder winters due to global climate change may therefore substantially increase pathogenic loads (Tylianakis et al., 2008). Insect vectors may also increase in population size, have more generations per year, and expand range limits (Pautasso et al., 2012). The general trend is likely to be a substantial increase in plant disease, but with some variability (Pautasso et al., 2012). Both plant and pathogen phenology will change under increased temperature. From a management viewpoint this can be important when considering the timing and efficacy of treatments (Chakraborty et al., 2000).

Sensitivity of pathogenic fungi to altered rainfall and extreme events

In general, dry weather favours insect vectors and viruses while wet weather favours fungi and bacteria (Anderson et al., 2004). This may lead to changes in the type and predictability of pathogen effects. Extreme events generally increase pathogen load due to increased plant stress (Anderson et al., 2004),

3.7 WILL CLIMATE CHANGE HAVE RELEVANT IMPACTS ON SOIL-PLANT-MICROBE SYSTEMS AND FEEDBACK EFFECTS ON SOIL SERVICES?

The previous sections show that climate change is likely to have direct effects on the abundance and composition of soil biota. These changes may in turn affect the processes that underpin ecosystem services. Climate change is also likely to have significant effects on plant community attributes, such as plant litter quality, plant community composition, and plant diversity. These changes indirectly affect processes mediated by the soil, and therefore the delivery of ecosystem services. As has been previously mentioned, the indirect effects to soil biota (particularly from changes to plant attributes) could potentially be larger than direct effects of climate change.

Here, we summarise previous research that is relevant to predicting how climate change impacts on plant and soil community attributes, and how subsequent changes to plant-soil feedbacks may affect the delivery of ecosystem services from New Zealand's primary sectors. We first describe how plant communities and plant behaviour may respond to climate change, and discuss the consequences of those changes for ecosystem services. Finally, we discuss how alterations to soil biotic communities may influence the provision of services. The discussion below is not always relevant to all sectors: the consequences of changes in plant productivity and litter quality are, whereas the consequences of changes in plant species richness and composition are more likely to be important for ecosystem services in forests and extensively grazed pastures than in sectors where plant species composition is more intensively managed, such as cropping and dairy pasture systems.

Plant community attributes

One of the key ways in which climate change may indirectly influence the provision of soil ecosystem services is by influencing the structure, biomass, and activity of the plant community. Among other effects, changes in plant attributes directly influence the C inputs and nutrients into soil (both from litter and root exudates), thus changes in plant community attributes are important to consider from the perspectives of soil C and nuteint cycling as well as soil biota.

Climate change may directly influence plant productivity through the CO_2 fertiliser effect, as the increase in CO_2 enhances the efficiency of photosynthesis. For example, eCO_2 has been found to increase crop productivity by up to 20%, and immature tree productivity by up to 30% (Tubiello et al., 2007). Warmer temperatures can increase growing season length, and combined with increased precipitation can increase nutrient availability (Davidson & Janssens, 2006), presumably by increased turnover of organic matter over a longer part of the year. Increased nutrient availability may also enhance plant productivity (Burke et al., 1997; Lukac et al., 2010). These positive effects may be limited in reality, however, as extreme events are likely to reduce plant productivity (Reyer, 2013), the CO_2 fertilisation effect may be reduced if other resources such as N or water become limiting (Saxe et al., 1998), and warmer temperatures may reduce soil water availability, which may in turn reduce responses. Although it is generally believed that eCO_2 will increase primary productivity, the net effect of all the aspects of climate change make the extent of this increase difficult to predict.

Climate change-driven increases in plant productivity, along with other plant behavioural responses, may alter soil-mediated processes through two main mechanisms: altered plant litter quality and increased below-ground C inputs. Increased productivity due to eCO_2 places greater demands on soil nutrient reserves, potentially leading to nutrient deficiencies that decrease the quality of plant litter (Saxe et al., 1998; Newton et al., 2010).

Immobilisation of nutrients in plant or soil organic pools (primarily N or P) can lead to progressive nutrient limitation (Luo et al., 2004). As previously mentioned, P may be limiting production as much as N at the NZFACE experiment (Gentile et al., 2012).

Climate change may also indirectly affect litter quality by influencing the environmental signals used to coordinate endogenous resource allocation in plants (Bazzaz et al., 1987). For example, eCO_2 has been associated with increased sugar and decreased N content in foliage, whereas increased temperature has induced the opposite effect (Curtis et al., 1998; Tingey et al., 2003).

Litter quality may also be altered by changes in how plants defend themselves from pests and diseases. The basal allocation of plant resources to defence when not threatened is largely dependent on the balance of C and nutrients available to the plant (Bazzaz et al., 1987). Climate change-driven alterations to C and nutrient availability are likely to alter the basal level of defence maintained by plants, potentially influencing the initial susceptibility and response of plants to pathogens and predation, and subsequently the quality of their litter. For example, improved resistance to fungal attack through greater lignification has been observed with increased temperature (Fuhrer, 2003), and high lignin contents are often indicative of lower litter quality (Berg, 2000). Resource availability will also have some degree of influence on the types of defence products plants synthesise in response to a pathogen attack, but given the complexity of plant secondary metabolism, uniform predictions of the effects of climate change on the production and activity of these molecules cannot be made (Bidart-Bouzat et al., 2008).

Climatic triggers and nutrient availability are also important signals in the synchronization of reproductive effort (Smaill et al., 2011), and it is therefore likely that climate change will induce alterations to the extent and frequency of the production of reproductive tissue, which is a source of high quality litter. Overall, it is likely that litter quality will be influenced by climate-induced changes in plant nutrient demand and availability, plant allocation to pathogen or herbivore defence, and the timing of reproductive effort.

Increases in productivity in response to climate change should also result in increased C inputs belowground, due to the greater production of photosynthate and increased turnover of plant biomass (Smith et al., 2008). For example, higher productivity may mean higher litter inputs to soil, and as plant demand for nutrients increases while C becomes more available, plants may allocate more C to belowground symbionts such as mycorrhizal fungi (Ceulemans et al., 1999; Rillig et al., 2000). Further, climate change may alter the relative allocation of resources to above and belowground plant biomass, and to the turnover rate of that biomass. For example, eCO_2 has been observed to stimulate allocation of resources to root biomass when under water stress, but not when under nutrient stress (Wang et al., 2010). Studies conducted under various land uses strongly indicates that eCO_2 increases below-ground C inputs through greater root biomass and turnover (Kimball et al., 2007; Lichter et al., 2008; Peralta & Wander, 2008), although the extent of the increase may be limited by the availability of N (Pregitzer et al., 2000). Examination of responses to increased temperature suggests decreased biomass allocation to roots is likely in some species (Way & Oren, 2010), but this is not consistently observed and is liable to be overcome by the effects of eCO_2 (Dieleman et al., 2012).

These predicted changes in C inputs to soil and litter quality have immediate implications for the saprotrophic and decomposer communities directly supported by plant litter, root exudates, and other plant-derived substances. The increased exudation of labile C caused by eCO_2 may facilitate the degradation of more complex C in soil organic matter, and the loss of soil C, by priming microbial activity (Drake et al., 2011). A counter argument to this position has been made based on the concept that increased inputs of labile C will cause the soil microbial community to shift to preferentially degrade this material, therefore reducing the rate at which litter and other more complex forms of C are decomposed, and reducing CO_2 emissions from these sources (Singh et al., 2010). Results from FACE studies in forest and pasture demonstrate increased losses of C from soil organic matter can occur with eCO_2 (Van Kessel et al., 2000; Gielen et al., 2005), providing support for the priming effect, although further work is needed to determine the longevity of this effect. Other studies provide support for the concept that activity relating to the degradation of labile substrates will be supported over that of the degradation of more recalcitrant substrates (He et al., 2010). Changes in litter quality will also have some influence on the structure of saprotrophic and decomposer communities and decomposition rates (Pritchard, 2011), but a synthesis of several studies suggests that the significance of any climate-change induced effects on the decomposer will be minimal, at least for eCO_2 alone (Norby et al., 2001).

Although climate change-induced changes in plant productivity and litter quality are likely to elicit some alterations to saprotrophic and decomposer communities and activity, the network of dependencies between plants and the various trophic and functional groups present in soil suggest that the wider effects of changes in C and litter inputs could be more significant (Pritchard, 2011). Changes in resource supply, particularly C availability, will substantially affect the flow of energy into soil and the relationships between plants and soil biota. These changes are likely to cascade through multiple trophic levels within the soil, with unpredictable consequences for plant metabolism, above- and belowground community composition, ecosystem productivity and agricultural yields (Pritchard, 2011). Although some predictions can be made for individual aspects of climate change, the extent and consistency of the response of C inputs and litter quality to the combined components of climate change is largely unclear. Most studies have considered short-term responses to variations in a single factor (e.g., CO_2) known to have leverage over C or litter inputs (Smith et al., 2008). Longer term studies of multiple factors, accounting for interactions and offsetting effects, are required to identify the ongoing changes in plant C and litter inputs resulting from climate change, and the response of the soil community.

As well as individual changes in plant C and nutrient dynamics, it is likely that the composition and diversity of plant communities will change in response to climate change, with potential consequences for soil-mediated ecosystem services. The positive effect of eCO_2 on productivity is generally found to be stronger for C3 plants compared to C4 plants, whereas warming is thought to benefit C4 plants over C3 plants. Legumes have also been found to respond positively to eCO_2 (Patterson et al., 1999, Fuhrer, 2003; Tubiello et al., 2007; Jaggard et al., 2010), unless nutrients are limiting (see discussion on Rhizobia at the NZFACE site). The direct effects of climate change on environmental conditions may increase the susceptibility of ecosystems to invasion, as invasive species are thought to be well-adapted to take advantage of a wide range of conditions, especially those associated with disturbance (Davidson et al., 2011). Any of these direct plant responses to climate change may alter competitive balances and therefore the relative abundance of existing species, and potentially the number of species, in both forest understories and extensively grazed systems.

As an example of how plant species composition may change in response to climate change, a recent grassland study found that the relative abundance of grasses declined with warming and warming combined with drought, whereas the relative abundance of legumes increased with warming (Cantarel et al., 2013). In forest systems, understory legumes may become more abundant, as long as other resources such as phosphorus and water do not become limiting (Reverchon et al., 2012). Various FACE studies in forests have also found that some understory species increase their relative growth rates (Mohan et al., 2007), and that the relative abundance of understory woody species (as opposed to herbaceous species) can increase (Souza et al., 2010).

As well as direct effects on the existing plant community, climate change may alter plant species composition and diversity indirectly, via changes to the abundance of herbivorous insects and microbial pathogens, and the effectiveness of biocontrol agents. The range, abundance, fecundity, and activity of insects and microbial plant pathogens are predicted to be altered by climate change (Patterson et al., 1999; Garrett et al., 2011). In particular, warmer temperatures may result in increased pest abundance and increased ranges, as insects will be able to over-winter more frequently and/or incorporate more generations per year (Patterson et al., 1999; Fuhrer, 2003). Extreme events such as drought may increase the incidence of insect outbreaks (Fuhrer, 2003). Other aspects of climate change, however, may reduce some pests and diseases. For example, drier summers may reduce pathogen abundance (Patterson et al., 1999). These potential outcomes also apply to biocontrol agents that are currently used to control undesirable plants or insects (Patterson et al., 1999; Colagiero et al., 2011). The effectiveness of pests and disease will also be modified by climate-change induced alterations to the production of defensive compounds by the host plant, as discussed above (Garrett et al., 2011).

Changes in plant behaviour will also affect the degree to which pests and disease impact on plant species. The stimulation of resource allocation to root biomass in response to eCO_2 has the potential to increase the activity of root predating insects (Johnson & McNicol, 2010). For example, one study found that eCO_2 resulted in enhanced nodulation of white clover, and a subsequent increase in the abundance of the larvae of the clover root weevil (*Sitona lepidus*; Johnson & McNicol 2010). Extended growing seasons or elongation of a developmental phase where plant disease susceptibility is enhanced may allow for greater pest and pathogen abundance as the window for predation and/or infection is expanded (Garrett et al., 2011).

Any climate change-driven disruption to the synchronization of plant and pest/pathogen behaviour is also likely to result in changes in the relative effect on plant health and survival. Alterations in the temporal development of plants relative to populations of above-ground herbivores have been found to induce various outcomes for plant performance (Pritchard, 2011), but it is largely unknown how below-ground pest and pathogen populations will respond. These climate change-induced changes in the effectiveness of pests, diseases, and biocontrol agents combined with the differential direct responses of plant species to climate change are likely to have consequences for the relative abundance of plant species and the species richness of plant communities.

Climate change-induced changes in plant community structure may affect soil biota and the supporting and regulating services they provide. Different plant species have been shown to have significant effects on several aspects of soil systems, such as pH, N mineralisation rates, phosphatase activity, and the structure of the soil microbial community (Bardgett et al., 1999; Orwin & Wardle, 2005; Orwin et al., 2010), all of which modify nutrient cycling in soil. These effects have been linked to plant traits, with plants that have fast growth rates and high quality litter being associated with bacteria-dominated microbial communities and high rates of N cycling (Tilman & Wedin, 1991; van der Krift & Berendse, 2001; Orwin et al., 2010). Litter decomposition rates have also been successfully linked to plant chemical traits such as C:N ratios and lignin contents, and leaf structural traits such as leaf dry matter content and specific leaf area (Garnier et al., 2004; Quested et al., 2007; Cornwell et al., 2008). A shift in plant communities towards those that produce lower quality litter should therefore reduce decomposition rates and potentially increase soil C storage. For example, an increase in low-productivity species in response to warming in a meadow ecosystem reduced C storage in soils initially as a result of reduced C inputs to the soil, but was predicted to increase soil C in the long term as a result of reduced decomposition rates in response to reduced litter quality (Saleska et al., 2002). Added to this, a recent study showed that individual plant species responses to warming, eCO_2 and drought in a mixed-plant community influenced the expression of soil microbial enzymes involved in P, S and C cycling (Kardol et al., 2010). This provides direct evidence of plant species modulating microbial responses to climate change.

As well as changes in the species composition of plant communities, changes in their species richness and evenness (the degree to which a community is dominated by one or more species) may also alter the delivery of ecosystem services. Plant species richness and/or diversity (i.e. the number of species present) has been shown to increase plant productivity (Hooper et al., 2005), which, as shown above, can have significant effects on belowground processes. More diverse communities can also result in greater nutrient retention in plant biomass (Hooper et al., 2005), with potential consequences for litter quality and microbial nutrient cycling. Plant species richness can also affect how plant community biomass responds to disturbances such as drought (van Ruijven & Berendse, 2010), which is also likely to influence how soil biota and processes respond. Recent evidence also suggests that subtle changes in species evenness influences ecosystem services such as nutrient cycling, leaching, and CO_2 fluxes (Orwin et al., unpublished data; Maestre et al., 2012). The extent to which plant species richness and evenness will interact with aspects of climate change other than drought is currently poorly investigated. However, a grassland warming experiment has showed that warming counteracted the positive effect of species richness on root biomass (De Boeck et al., 2007), and that plant species richness only had a positive effect on potential nitrification rates under warmed conditions (Malchair et al., 2010). A further study found that the eCO_2 increased earthworm cast production in a model calcareous grassland, but only at high plant species richness (Arnone et al., 2013). These results suggest that climate change-induced changes in plant species richness and composition may have significant influences on soil-mediated processes that underpin ecosystem services.

Soil biotic community attributes

Changes in plant communities, along with the direct effects of climate change, are likely to affect the biomass, diversity, and composition of soil biotic communities (see details in Soil Biota and section above). For example, it is expected that eCO_2 will alter the bacterial:fungal ratio and increase saprotrophic biomass and mesofauna (Blankinship et al., 2011; Pritchard, 2011). Interactions of eCO_2 with other climate change factors and the indirect effects of changes in plant communities and their behaviour may modify these responses, making it difficult to predict how soil biota will respond to overall climate change (Kardol et al., 2010; Blankinship et al., 2011). Although we do not currently know the specific details of which soil species within communities will change most in response to climate change, we can use the general ecological literature in this area to assess whether changes in biomass, diversity, and composition may have flow-on effects to the provision of ecosystem services. Changes in soil biotic community attributes are likely to be relevant to all sectors.

Species richness, or the number of species in a community, may change in response to climate change. There is considerable debate as to whether the species richness of soil biota is likely to influence ecosystem processes (Wurst et al., 2012). For soil microbes in particular, this is due to the incredibly high number of species found in any one soil and because many processes can be performed by a large number of those species, which is thought to lead to functional redundancy (Hooper et al., 2005). However, there is some evidence that soil biotic diversity can influence functioning. The species richness of mycorrhizal fungi has been shown to influence plant productivity (van Der Heijden et al., 1998; Jonsson et al., 2001) via increased nutrient uptake (van Der Heijden et al., 1998). Two meta-analyses of over 20 studies have also found that soil biotic diversity can have positive effects on C cycling (Srivastava et al., 2009; Nielsen et al., 2011), and the stability of some processes (e.g., denitrification) may be higher in more diverse systems (Hallin et al., 2012). However, many studies have used unrealistic gradients of species richness, i.e. unrealistically low species numbers that would not be found under natural conditions (Gessner et al., 2010; Nielsen et al., 2011), making it difficult to extrapolate results to the real world. When this is taken into consideration along with other studies that show that even the loss of 99.99% of the soil's microbial diversity does not reduce C mineralisation, nitrification or denitrification rates (Wertz et al., 2006), and a general lack of studies that look at diversity across trophic levels, the consequences for ecosystem services of any change in species richness of soil biota in response to climate change remains unclear (Wurst et al., 2012).

In contrast to species richness effects, there is greater certainty that the composition of soil biota is important. Mycorrhizal species are known to have differential effects on plant community composition, plant water use, root biomass, and plant productivity (Rillig & Mummey, 2006), with potential consequences for the delivery of ecosystem services such as nutrient retention and food production. For example, research with Scots pine and a selection of ectomycorrhizal fungi suggests that the development of associations with particular mycorrhizal species can reduce the incidence of nutrient deficiencies by enabling access to a broader pool of soil nutrients, allowing climate change-driven productivity increases to be realised (Gorissen et al., 2000) and litter quality to remain constant. Different mycorrhizal species also differ in their architecture and hyphal production rate (Rillig & Mummey, 2006), which may influence soil structure and subsequently soil process rates.

Mycorrhizal associations may also be particularly important in acquiring phosphorus (P) when it is limiting. Since P can often be the limiting nutrient (or co-limiting with N), and P cycling has not been explored in other sections, we do so briefly here. The P cycle differs somewhat from other nutrient cycles (Condron et al., 2005) in that in systems where P has not been added, the largest form of available P is often in the organic form. The release of plant-available P from soil minerals is slow, and some soils (Allophanic Soils in particular) strongly sorb available P. Therefore, P is less likely to leach than other nutrients, but is susceptible to loss from surface erosion.

In many systems, the availability of P is determined by the mineralisation of the organic material. Just as with N, the increased plant growth from eCO_2 can result in P limitation (Finzi et al., 2010), but this may be less of a problem in agricultural soils where additional P has been added as fertiliser. There have been far fewer studies on P (as opposed to N) dynamics during climatic change. In a meta-study, Menge and Field (2007) suggest that only when there are increases in NPP (often indirectly through N addition) does climate change have an effect on decreasing P availability. In such situations, specific mycorrhizal associations that maximise P uptake to the host plant may provide a competitive advantage over other plants that do not posses similar associations.

Mycorrhizal fungi also vary in the quality and quantity of C they produce. As much of this C enters the pools with slower turnover rates in the soil, changes in mycorrhizal species composition may alter the amount of C stored in ecosystems (Treseder & Allen, 2000; Orwin et al., 2011). On the other hand, recent evidence also suggests that, under eCO_2, the presence of arbuscular mycorrhizal fungi might increase the decomposition of organic matter through release of easily decomposable material (i.e., the priming effect where release of easily decomposable material stimulates the decomposition of more recalcitrant material), and so increase C loss from soils (Cheng et al., 2011). Mycorrhizal fungal composition, along with the composition of other soil fungi, may also influence soil water repellency as different species produce different hydrophobic proteins (Smits et al., 2003; Rillig, 2005; Rillig & Mummey, 2006).

The community composition of other soil biota has also been shown to be important for ecosystem processes. For example, denitrifier bacterial identity can influence denitrification rates, including under climate change scenarios (Salles et al., 2009; Cantarel et al., 2012), soil microbial community structure can influence C (Strickland et al., 2009; Nielsen et al.m 2011), and N cycling rates (Balser & Firestone, 2005), and the composition and traits of macrofauna species can explain much of the variation in litter decomposition (Heemsbergen et al., 2004). Soil pathogens can also affect plant growth significantly, but effects tend to be species-specific (Kulmatiski et al., 2008; Maron et al., 2011). Increases in the fungal:bacterial ratio are associated with increased nutrient retention (including when there are wet–dry cycles), slower nutrient cycling and higher soil C (Wardle et al., 2004; de Vries & Bardgett, 2012). Further, priming effects were found to be lower where fungi and actinomycete abundance was reduced and composition altered by fumigation (Garcia-Pausas & Paterson, 2011).

Climate change can also influence the expression of genes within microbial communities. For example, Zhou et al. (2012) found that the expression of genes involved in the degradation of labile substrates, denitrification, and N fixation were enhanced by warming, whereas those that are involved in degrading more recalcitrant substrates were not. These results suggest that changes in the composition and activity of soil biotic communities in response to climate change are likely to have a significant effect on ecosystem services (Wurst et al., 2012).

It is also likely that the biomass of soil biota will change in response to climate change. Increases in biomass may result in increased levels of functioning. Evidence for this view-point comes from studies where, for example, bacterial biomass was positively correlated to N fluxes and soil respiration (Cantarel et al., 2012), increased microbial biomass was associated with increases in enzyme activities under eCO_2 (Drissner et al., 2007) and meta-analyses that suggest increases in micro- and mesofauna abundance can increase above-ground plant productivity (Sackett et al., 2010).

Effects may be system specific, however. For example, increases in soil fauna were found to have much stronger effects on plant productivity in systems where N is limiting (e.g., coniferous forests) compared with systems that contained legumes (Sackett et al., 2010).

Also, as has been previously discussed, greater numbers of specific biotic groups (such as root-feeding nematodes), may decrease productivity. Trophic cascades (the alteration of lower trophic levels by a key predator, such as suppression of prey numbers) may limit the extent to which increases or decreases in biomass affect ecosystem services (Wardle, 2002). Overall, it is clear that changes in the biomass and composition, and possibly the species richness, of soil biota, could have a significant effect on the delivery of ecosystem services. However, it is difficult to predict what direction these changes in soil biotic communities will take, and what their consequences will be.

Greenhouse gas emissions

Given that soil microorganisms regulate terrestrial greenhouse gas (CO_2, CH_4 and N_2O) flux, there is significant potential for feedback interactions to occur as climate change alters conditions within soil, but there is still much dispute over the extent and longevity of any effect.

A principle factor driving increased emissions is the sensitivity of soil microbial metabolism to temperature. Increased temperature increases the rate of respiration more than photosynthesis, suggesting a net increase in CO_2 production is likely to occur in the soil-plant system (Woodwell et al., 1998). Attempts to predict the extent of the change in microbial respiration are complicated by differences in the forms and stability of C pools in soil, but integrated models suggest that increased emissions, albeit over varying timescales, are the most likely outcome (Knorr et al., 2005). The increased exudation of labile C is also likely to stimulate respiration by enhancing the decomposition of more complex soil organic matter, as discussed previously (Van Kessel et al., 2000; Gielen et al., 2005; Drake et al., 2011).

The long-term effects on microbial respiration, however, will be significantly influenced by the extent of any nutrient limitations that may occur as plants capture greater volumes of nutrients to support increased biomass production (Singh et al., 2010). Such limitations are likely to cause shifts to fungal-dominated systems that are inherently low respiration. The role of soil water is also debated as increased moisture availability may stimulate respiration in some regions but decrease oxygenation in others, whereas less moisture will decrease microbial metabolism but increase oxygenation in existing swamps and peat lands (Singh et al., 2010). This will have a major influence on nitrous oxide (N_2O) production and emission from soils: more discussion on climate change impacts on N_2O emissions can be found in section 5.4.

Alterations to CH_4 dynamics in soil are of great importance because CH_4 has approximately 25 times the global warming potential of CO_2 on a molecular basis, and other than atmospheric chemical oxidation, is only captured by methanotrophs in aerobic soils (Le Mer & Roger, 2001). Increased temperature is likely to stimulate methane emissions by increasing rates of microbial metabolic activity (Woodwell et al., 1998; Singh et al., 2010), but the determining factor in the long term will likely be the effects of climate change on the activity of methanotrophs. Based on comparisons of climatic conditions across various regions it is highly probable that CH_4 consumption in wetter soils will decrease due to reduced oxygenation. Elevated CO_2 has the potential to reduce CH_4 uptake substantially, but the mechanisms for this are not understood (Dalal et al., 2008; Singh et al., 2010). While there is likely to be some inhibitory effects of moisture and eCO_2 on methanogenic soil bacteria, there is no reason to assume this will match the substantial inhibition of methanotrophs that has been reported. Consequently, it is reasonable to assume that climate change will stimulate decreased CH_4 consumption in soils, producing a net increase in emissions.

Combined, these results suggest that the effects of climate change on ecosystem services that are mediated by plants, soil biota and the interactions between them are likely to depend on both individual and community-level responses. Individual responses at the organismal level may be relatively rapid and occur within a few years, while changes in community structure may occur over years to many decades (Bardgett et al., 2013). This means that the overall effect of climate change on the delivery of ecosystem services will change over time. Any decoupling caused by differential responses of plant and soil communities to climate change may have significant consequences for the processes they mediate, an effect which is more likely over longer time scales as more major changes in community structure occur due to the immigration and loss of species (Bardgett et al., 2013).

3.8 REFERENCES

A'Bear, A.D., Boddy, L., Hefin, J.T. (2012). Impacts of elevated temperature on the growth and functioning of decomposer fungi are influenced by grazing Collembola. *Global Change Biology 18,* 1823-1832.

Allard V., Newton P.C.D., Lieffering M., Soussana J.F., Carran, R.A., Matthew, C. (2004). Elevated CO_2 effects on decomposition processes in a grazed grassland. *Global Change Biology 10,* 1553-1564.

Allard V., Newton P.C.D., Lieffering M., Soussana J.F., Carran, R.A., Matthew, C. (2005). Increased quantity and quality of coarse soil organic matter fraction at elevated CO_2 in a grazed grassland are a consequence of enhanced root growth rate and turnover. *Plant and Soil 276,* 49-60.

Allard, V., Robin, C., Newton, P.C.D., Lieffering, M., Soussana, J.F. (2006). Short and long-term effects of elevated CO_2 on Lolium perenne rhizodeposition and its consequences on soil organic matter turnover and plant N yield. *Soil Biology & Biochemistry 38,* 1178-1187.

Anderson, P.K., Cunningham, A.A., Patel, N.G., Morales, F.J., Epstein, P.R., Daszak, P. (2004). Emerging infectious diseases of plants: pathogen pollution, climate change and agrotechnology drivers. *Trends in Ecology and Evolution 19,* 535-544.

Andrews, J.A., Schlesinger, W.H. (2001). Soil CO_2 dynamics, acidification, and chemical weathering in a temperate forest with experimental CO_2 enrichment. *Global Biogeochemical Cycles 15,* 149-162.

Antoninka, A., Reich, P.B., & Johnson, N.C. (2011). Seven years of carbon dioxide enrichment, nitrogen fertilization and plant diversity influence arbuscular mycorrhizal fungi in a grassland ecosystem. *New Phytologist 192,* 200-214.

Arnone, J.A., Zaller, J.G., Hofer, G., Schmid, B., Korner, C. (2013). Loss of plant diversity eliminates stimulatory effect of elevated CO_2 on earthworm activity in grasslands. *Oecologia 171,* 613-622.

Avrahami, S., Liesack, W., & Conrad, R. (2003). Effects of temperature and fertilizer on activity and community structure of soil ammonia oxidizer. *Environmental Microbiology 5,* 691-705.

Ayres, E., Wall, D.H., Simmons, B.L., Field, C.B., Milchunas, D.G., Morgan, J.A., Roy J. (2008). Belowground nematode herbivores are resistant to elevated atmospheric CO_2 concentrations in grassland ecosystems. *Soil Biology & Biochemistry 40,* 978-985.

Bakonyi, G., Nagy, P., Kovács-Láng, E., Kovács, E., Barabás, S., Répási, V., Seres, A. (2007). Soil nematode community structure as affected by temperature and moisture in a temperate semiarid shrubland. *Applied Soil Ecology 37,* 31-40.

Balser, T.C., Firestone, M.K. (2005). Linking microbial community composition and soil processes in a California annual grassland and mixed-conifer forest. *Biogeochemistry 73,* 395-415.

Bapiri, A., Bääth, E., Rousk, J. (2010). Drying-rewetting cycles affect fungal and bacterial growth differently in an arable soil. *Microbial Ecology 60,* 419-428.

Bardgett, R.D., Freeman, C., Ostle, N.J. (2008). Microbial contributions to climate change through carbon cycle feedbacks. *ISME Journal 2,* 805-814.

Bardgett, R.D., Kandeler, E., Tscherko, D., Hobbs, P.J., Bezemer, T.M., Jones, T.H., Thompson, L.J. (1999). Below-ground microbial community development in a high temperature world. *Oikos 85,* 193-203.

Bardgett, R.D., Manning, P., Morrien, E., de Vries, F.T. (2013). Hierarchical responses of plant soil interactions to climate change:consequences for the global carbon cycle. *Journal of Ecology 101,* 334-343.

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

Bazzaz, F.A., Chiariello, N.R., Coley, P.D., Pitelka, L.F. (1987). Allocating resources to reproduction and defense. *BioScience 37,* 58-67.

Beed, F., Benedetti, A., Cardinali, G., Chakraborty, S., Dubois, T., Garrett, K., Halewood, M. (2011). Climate change and micro-organism genetic resources for food and agriculture: state of knowledge, risks and opportunities. Background study paper no. 57. Commission on Genetic Resources for Food and Agriculture, 66 pp.

Bell, N.L., Knight, K.W.L., Mercer, C.F., Shah, F., Sturhan, D., Aalders, L.T., Yeates, G.W., Watson, R.N., Marshall, J.W., Page, G.D. (2006). Occurrence and distribution of Meloidogyne spp. nematodes in New Zealand. *Journal of Nematology 38,* 261 (Abstract).

Bell, N.L., Watson, R.N. (2001). Identification and host range assessment of the nematodes *Paratylenchus nanus* (Tylenchida: Tylenchulidae) and *Paratrichodorus minor* (Triplonchida: Trichodoridae). *Nematology 3,* 483-490.

Bergner, B., Johnstone, J., Treseder, K.K. (2004). Experimental warming and burn severity alter soil CO_2 flux and soil functional groups in a recently burned boreal forest. *Global Change Biology 10,* 1996-2004.

Bernard, E.C., Jennings, P.L. (1997). Host range and distribution of the clover root-knot nematode, Meloidogyne trifoliophila. *Journal of Nematology 29,* 662-672.

Bidart-Bouzat, M.G., Imeh-Nathaniel, A. (2008). Global change effects on plant chemical defenses against insect herbivores. *Journal of Integrative Plant Biology 50*, 1339-1354.

Blankinship, J.C., Niklaus, P.A., Hungate, B.A. (2011). A meta-analysis of responses of soil biota to global change. *Oecologia 165*, 553-565.

Boag, B., Geoghegan, I.E. (1984). An evaluation of agricultural and horticultural crops as hosts for the plant-parasitic nematode Longidorus elongatus (Nematoda, Dorylaimida). *Crop Research 24,* 85-95.

Boisvenue, C., Running, S.W. (2006). Impacts of climate change on natural forest productivity - Evidence since the middle of the 20th century. *Global Change Biology 12*(5), 862-882.

Bowatte, S., Carran, R.A., Newton, P.C.D., Theobald, P. (2008). Does atmospheric CO_2 concentration influence soil nitrifying bacteria and their activity? *Australian Journal of Soil Research 46,* 617-622.

Brady, P.V., Carroll, S.A. (1994). Direct effects of CO_2 and temperature on silicate weathering: Possible implications for climate control. *Geochimica et Cosmochimica Acta 58*, 1853-1856.

Brinkman R., Sombroek W.G. (1996). The effects of global change on soil conditions in relation to plant growth and food production. In: Bazzaz F., Sombroek W., (Eds) *Global climate change and agricultural production. Direct and indirect effects of changing hydrological, pedological and plant physiological processes*. FAO, Rome, FAO and John Wiley & Sons, New York.

Brinkman, R. (1982). Clay transformations: aspects of equilibrium and kinetics. In G.H. Bolt (Ed.), *Soil Chemistry. B. Physicochemical Models. Developments in Soil Science 5B. 2nd ed.* (pp. 433-458) Elsevier, Amsterdam.

Briones, M.J.I., Ostle, N.J., Garnett, M.H. (2007). Invertebrates increase the sensitivity of non-labile soil carbon to climate change. *Soil Biology and Biochemistry 39*, 816-818.

Bristow K.L., Marchant S.M., Deurer M., Clothier B.E. (2010). Soil – A key component of ecological infrastructure. Proceedings of the 19th World Congress of Soil Science, Brisbane, Australia, 1–7 August 2010 (on CD).

Bryla, D., Eissenstat, D. (2005). Respiratory costs of mycorrhizal associations. Plant Respiration 207–224.

Bulgarelli, D., Rott, M., Schlaeppi, K., van Themaat, E.V.L., Ahmadinejad, N., Assenza, F., Rauf, P., Huettel, B., Reinhardt, R., Schmelzer, E., Peplies, J., Gloeckner, F.O., Amann, R., Eickhorst, T., Schulze-Lefert, P. (2010). Revealing structure and assembly cues for Arabidopsis root-inhabiting bacterial microbiota. *Nature 488,* 91-95.

Cantarel, A.A.M., Bloor, J.M.G., Pommier, T., Guillaumaud, N., Moirot, C., Soussana, J.F., Poly, F. (2012). Four years of experimental climate change modifies the microbial drivers of N_2O fluxes in an upland grassland ecosystem. *Global Change Biology 18,* 2520-2531.

Cantarel, A.A.M., Bloor, J.M.G., Soussana, J.F. (2013). Four years of simulated climate change reduces above-ground productivity and alters functional diversity in a grassland ecosystem. *Journal of Vegetation Science 24,* 113-126.

Castro, H.F., Classen, A.T., Austin, E.E., Norby, R.J., Schadt, C.W. (2010). Soil microbial community responses to multiple experimental climate change drivers. *Applied and Environmental Microbiology 76,* 999-1007.

Cavigelli, M.A., Robertson, G.P. (2000). The functional significance of denitrifier community composition in a terrestrial ecosystem. *Ecology 81,* 1402-1414.

Chakraborty, S., Tiedemann, A.V., Teng, P.S. (2000). Climate change: potential impact on plant diseases. *Environmental Pollution 108,* 317-326.

Cheng, L., Booker, F.L., Burkey, K.O., Tu, C., Shew, D., Rufty, T.W., Fiscus, E.L., Deforest, J.L., Hu, S. (2011). Soil microbial responses to elevated CO_2 and O_3 in a nitrogen-aggrading agroecoystem. *PLoS ONE, 6,* e21377. doi:10.1371/journal.pone.0021377

Cheng, L., Booker, F.L., Tu, C., Burkey, K.O., Zhou, L., Shew, H.D., Rufty, T.W., Hu, S. (2012). Arbuscular mycorrhizal fungi increase organic carbon decomposition under elevated CO_2. *Science 337,* 1084-1087.

Chevallier, A.J.T., Lieffering, M., Carran, R.A., Newton, P.C.D. (2006). Mineral nitrogen cycling through earthworm casts in a grazed pasture under elevated atmospheric CO_2. *Global Change Biology 12,* 56-60.

Cichota R., Vogeler I., Snow V., (2010). Describing N leaching under urine patches in pastoral soil. Proceedings of the 19th World Congress of Soil Science, Brisbane, Australia, 1–7 August 2010 (on CD).

Clark A.J., Nottage R.A.C., Wilcocks L., Lee J.M., Burke C., Kalaugher E., Roche J., Beukes P., Lieffering M., Newton P.C.D., Li F.Y., Vibart R., Teixeira E.I., Brown H.E., Fletcher A.L., Hernandez-Ramirez G., Soltani A., Viljanen-Rollinson S., Horrocks A., Johnstone P., Clothier B., Hall A., Green S., Dunningham A., Kirschbaum M.U.F., Meason D., Payn T., Collins D.B.G., Woods R.A., Rouse H., Duncan M., Snelder T., Cowie B., (2012). Impacts of climate change on land-based sectors and adaptation options. In: Clark A.J., Nottage R.A.C. (Eds.) *Impacts of climate change on land-based sectors and adaptation options: Technical Report to the Sustainable Land Management and Climate Change Adaptation Technical Working Group,* Ministry for Primary Industries. 408 p.

Clothier B., Hall A., Green S., (2012). Horticulture: Adapting the horticultural and vegetable industries to climate change. In: Clark A.J., Nottage R.A.C. (Eds.) *Impacts of climate change on land-based sectors and adaptation options: Technical Report to the Sustainable Land Management and Climate Change Adaptation Technical Working Group,* Ministry for Primary Industries. Pp. 237–292.

Clothier B.E., Green S.R., Deurer M., (2008). Preferential flow and transport in soil: progress and prognosis. *European Journal of Soil Science 59*, 2-13.

Cock, M.J.W., Biesmeijer, J.C., Cannon, R.J.C., Gerard, P.J., Gillespie, D., Jiménez, J.J., Lavelle, P.M., Raina, S.K. (2011). Climate change and invertebrate genetic resources for food and agriculture: state of knowledge, risks and opportunities. Background study paper no. 54. Commission on Genetic Resources for Food and Agriculture, 105 pp.

Colagiero, M., Ciancio, A. (2011). Climate changes and nematodes: Expected effects and perspectives for plant protection. *Redia 94,* 113-118.

Condron, L.M. and Tiessen, H. (2005). Interactions of organic phosphorus in terrestrial ecosystems. In: *Organic Phosphorus in the environment*. Eds., B.L Turner, E. Frossard and D. Baldwin. CAB Interrnational, Wallingford, UK pp. 295-307.

Cornwell, W. K., J. H. C. Cornelissen, K. Amatangelo, E. Dorrepaal, V. T. Eviner, O. Godoy, S. E. Hobbie, B. Hoorens, H. Kurokawa, N. Perez-Harguindeguy, H. M. Quested, L. S. Santiago, D. A. Wardle, I. J. Wright, R. Aerts, S. D. Allison, P. van Bodegom, V. Brovkin, A. Chatain, T. V. Callaghan, S. Diaz, E. Garnier, D. E. Gurvich, E. Kazakou, J. A. Klein, J. Read, P. B. Reich, N. A. Soudzilovskaia, M. V. Vaieretti, Westoby, M. (2008). Plant species traits are the predominant control on litter decomposition rates within biomes worldwide. *Ecology Letters 11,* 1065-1071.

Costanza, R., dArge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Raruelo, J., Raskin, R.G., Sutton, P., and van den Belt, M. (1997). The value of the world's ecosystem services and natural capital. *Nature,* 387, 253-260.

Cox P.M., Betts R.A., Jones C.D., Spall S.A., Totterdell I.J. (2000). Acceleration of global warming due to carbon cycle feedbacks in a coupled climate model. *Nature 408*, 184–187.

Curtis, P.S., Wang, X. (1998). A meta-analysis of elevated CO_2 effects on woody plant mass, form, and physiology. *Oecologia 113,* 299-313.

Curtis, T.P., Sloan, W.T., Scannell, J.W. (2002). Estimating prokaryotic diversity and its limits. *Proceedings of the National Academy of Sciences 99,* 10494-10499.

Davidson, A.M., Jennions, M., Nicotra, A.B. (2011). Do invasive species show higher phenotypic plasticity than native species and, if so, is it adaptive? A meta-analysis. *Ecology Letters 14,* 419-431.

Davidson, E.A., Janssens, I.A. (2006). Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. *Nature 440,* 165-173.

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

de Vries, F.T., Liiri, M.E., Bjørnlund, L., Bowker, M.A., Christensen, S., Setälä, H.M., Bardgett, R.D. (2012). Land use alters the resistance and resilience of soil food webs to drought. *Nature Climate Change 2*, 276-280.

DeBano, L.F. (1981) Water repellent soils: a state-of-the-art. USDA Forest Service Technical Report PSW-46. 21 p.

Deurer M., Grinev D., Young I., Clothier B.E., Müller K., (2009). The impact of soil carbon management on soil macro-pore structure: A comparison of two apple orchard systems in New Zealand. *European Journal of Soil Science 60,* 945–955.

Dickie, I.A. (2007). Host preference, niches and fungal diversity. *New Phytologist 174,* 230-233.

Dickie, I.A., Bolstridge, N., Cooper, J.A., Peltzer, D.A. (2010). Co-invasion by Pinus and its mycorrhizal fungi. *New Phytologist 187,* 475-484.

Dominati, E., Patterson, M., Mackay, A. (2010). A framework for classifying and quantifying the natural capital and ecosystem services of soils. *Ecological Economics 69,* 1858-1868.

Drissner, D., Blum, H., Tscherko, D., Kandeler, E. (2007). Nine years of enriched CO_2 changes the function and structural diversity of soil microorganisms in a grassland. *European Journal of Soil Science 58,* 260-269.

Edwards, E.J., McCaffery, S., Evans, J.R. (2006). Phosphorus availability and elevated CO_2 affect biological nitrogen fixation and nutrient fluxes in a clover-dominated sward. *New Phytologist 169,* 157-167.

Eisenhauer, N., Cesarz, S., Koller, R., Worm, K., Reich, P.B. (2012). Global change belowground: impacts of elevated CO_2, nitrogen, and summer drought on soil food webs and biodiversity. *Global Change Biology 18,* 435-447.

Elshahed, M.S., Youssef, N.H., Spain, A.M., Sheik, C., Najar, F.Z., Sukharnikov, L.O., Roe, B.A., Davis, J.P., Schloss, P.D., Bailey, V.L., Krumholz, L.R. (2008). Novelty and uniqueness patterns of rare members of the soil biosphere. *Applied and Environmental Microbiology 74*, 5422-5428.

Entry, J.A., Rygiewicz, P.T., Watrud, L.S., Donnelly, P.K. (2002). Influence of adverse soil conditions on the formation and function of Arbuscular mycorrhizas. *Advances in Environmental Research 7,* 123-138.

Fierer, N., Schimel, J.P., Holden, P.A. (2003). Influence of drying–rewetting frequency on soil bacterial community structure. *Microbial Ecology 45,* 63-71.

Fitter, A.H., Heinemeyer, A., Staddon, P.L. (2000). The impact of elevated CO_2 and global climate change on arbuscular mycorrhizas: a mycocentric approach. *New Phytologist 147,* 179-187.

Fitter, A.H., Helgason, T., Hodge, A. (2011). Nutritional exchanges in the arbuscular mycorrhizal symbiosis: Implications for sustainable agriculture. *Fungal Biology Reviews 25,* 68-72.

Flechard, C.R., Neftel, A., Jocher, M., Ammann, C., Leifeld, J., Fuhrer, J. (2007). Temporal changes in soil pore space CO_2 concentration and storage under permanent grassland. *Agricultural and Forest Meteorology 142*, 66-84.

Frey, S.D., Drijber, R., Smith, H., Melillo, J. (2008). Microbial biomass, functional capacity, and community structure after 12 years of soil warming. *Soil Biology and Biochemistry 40,* 2904-2907.

Frey, S.D., Elliott, E.T., Paustian, K. (1999). Bacterial and fungal abundance and biomass in conventional and no-tillage agroecosystems along two climatic gradients. *Soil Biology and Biochemistry 31,* 573-585.

Fuhrer, J. (2003). Agroecosystem responses to combinations of elevated CO_2, ozone, and global climate change. *Agriculture, Ecosystems and Environment 97,* 1-20.

Garcia-Pausas, J. Paterson, E. (2011). Microbial community abundance and structure are determinants of soil organic matter mineralisation in the presence of labile carbon. *Soil Biology & Biochemistry 43,* 1705-1713.

Garnier, E., Cortez, J., Billes, G., Navas, M.L. , Roumet, C., Debussche, M., Laurent, G., Blanchard, A., Aubry, D., Bellmann, A., Neill, C. Toussaint, J.P. (2004). Plant functional markers capture ecosystem properties during secondary succession. *Ecology 85,* 2630-2637.

Garrett, K.A., Dendy, S.P., Frank, E.E., Rouse, M.N., Travers, S.E. (2006). Climate change effects on plant disease: genomes to ecosystems. *Annual Review of Phytopathology 44,* 489-509.

Garrett, K.A., Forbes, G.A., Savary, S., Skelsey, P., Sparks, A.H., Valdivia, C., Van Bruggen, A.H.C., Willocquet, L., Djurle, A., Duveiller, E., Eckersten, H., Pande, S., Vera Cruz, C., Yuen, J. (2011). Complexity in climate-change impacts: An analytical framework for effects mediated by plant disease. *Plant Pathology 60,* 15-30.

Gentile, R., Dodd, M., Lieffering M., Brock S.C., Theobold, P.W., & Newton, P.C.D. (2012). Effects of long-term exposure to enriched CO_2 on the nutrient supplying capacity of a grassland soil. *Biology and Fertility of Soils 48,* 357-362.

Gessner, M.O., Swan, C.M., Dang, C.K., McKie, B.G., Bardgett, R.D., Wall, D.H. Hattenschwiler, S. (2010). Diversity meets decomposition. *Trends in Ecology & Evolution 25,* 372-380.

Gorissen, A., Kuyper, Th.W. (2000). Fungal species-specific responses of ectomycorrhizal Scots pine (Pinus sylvestris) to elevated [CO_2]. *New Phytologist 146*, 163-168.

Goudie, A.S., Viles, H.A. (2012). Weathering and the global carbon cycle: Geomorphological perspectives. *Earth Science Reviews 113,* 59-71.

Green S., Deurer M., Clothier B., Andrews S., Cauldwell K., Roberts A., Wellwood M., Thomas P. (2010). Water and nitrogen movement under agricultural and horticultural land. *Proceedings of the*

Workshop 'Farming's Future: Minimising footprints and maximising margins'. Fertiliser & Lime Research Centre, Massey University, 10–11 February 2010.

Greer G., Saunders, C. (2012). Agribusiness and Economics Research Unit Report.

Grey, S.B., Classen, A.T., Kardol, P., Yermakov, Z., Miller, R.M. (2011). Multiple climate change factors interact to alter soil microbial community structure in an Old-field ecosystem. *Soil Science Society of America Journal 75*, 2217-2226.

Hallin, S., Welsh, A., Stenstrom, J., Hallet, S., Enwall, K., Bru, D., Philippot, L. (2012). Soil Functional Operating Range Linked to Microbial Biodiversity and Community Composition Using Denitrifiers as Model Guild. *Plos One* 7, e51962.

Hayden, H.L., Mele, P.M., Bougoure, D.S., Allan, C.Y., Norng, S., Piceno, Y.M., Brodie, E.L., DeSantis, T.Z., Andersen, G.L., Williams, A.L., Hovenden, M.J. (2012). Changes in the microbial community structure of bacteria, archaea and fungi in response to elevated CO_2 and warming in an Australian native grassland soil. *Environmental Microbiology 14*, 3081-3096.

Hendrix, P.F., Parmelee, R.W., Crossley, D.A., Coleman, D.C., Odum, E.P., Groffman, P.M. (1986). Detritus food webs in conventional and non-till agroecosystems. *BioScience 36*, 374-380.

Holmstrup, M., Sørensen, J.G., Maraldo, K., Schmidt, I.K., Mason, S., Tietema, A., Smith, A.R., Emmett, B., Schmelz, R.M., Bataillon, T., Beier, C., Ehlers, B.K. (2012). Increased frequency of drought reduces species richness of enchytraeid communities in both wet and dry heathland soils. *Soil Biology and Biochemistry 53*, 43-49.

Hooper, D.U., Chapin, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, M., Naeem, S., Schmid, S., Setala, H., Symstad, A.J., Vandermeer, J., Wardle, D.A. (2005). Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. *Ecological Monographs 75*, 3-35.

Horz, H.P., Barbrook, A., Field, C.B., Bohannan, B.J.M. (2004). Ammonia oxidising bacteria respond to multi-factoral global change. *PNAS 101*, 15136-15141.

Hu, S., Tu, C., Chen, X., Gruver, J.B. (2006). Progressive N limitation of plant response to elevated CO_2: a microbiological perspective. *Plant and Soil 289*, 47-58.

Hu, S., Wu, J., Burkey, K.O., Firestone, M.K. (2005). Plant and microbial N acquisition under elevated atmospheric CO_2 in two mesocosm experiments with annual grasses. *Global Change Biology 11*, 213-223.

Hunt, H.W., Coleman, D.C., Ingham, E.R., Ingham, R.E., Elliott, E.T., Moore, J.C., Rose, S.L., Reid, C.P.P., Morley, C.R. (1987). The detrital food web in a shortgrass prairie. *Biology and Fertility of Soils 3*, 57-68.

Izaurralde, R.C., Thomson, A.M., Morgan, J.A., Fay, P.A., Polley, H.W., Hatfield, J.L. (2011). Climate impacts on agriculture: Implications for forage and rangeland production. *Agronomy Journal 103*, 371-381.

Jaggard, K.W., Qi, A., Ober, S. (2010). Possible changes to arable crop yields by 2050. *Philosophical Transactions of the Royal Society B: Biological Sciences 365*, 2835-2851.

Jeyakumar P., Müller K., Deurer M., Van den Dijssel C., Mason K., Blackburn R., Green S., Clothier B., (2012) *Soil water repellency – Economic and environmental consequences of an emerging issue of soil degradation*. A Final Report by Plant & Food Research to AGMARDT. 46 p.

Jonsson, L.M., Nilsson, M.C., Wardle, D.A., Zackrisson, O. (2001). Context dependent effects of ectomycorrhizal species richness on tree seedling productivity. *Oikos 93*, 353-364.

Johnson, S.N., McNicol, J.W. (2010). Elevated CO_2 and aboveground-belowground herbivory by the clover root weevil. *Oecologia 162*, 209-216.

Kandeler, E., Tscherko, D., Bardgett, R.D., Hobbs, P.J., Kampichler, C., Jones, T.H. (2002). The response of soil microorganisms and roots to elevated CO_2 and temperature in a terrestrial model ecosystem. *Plant and Soil 202*, 251-262.

Kardol, P., Cregger, M.A., Campanv, C.E., Classen, A.T. (2010). Soil ecosystem functioning under climate change: Plant species and community effects. *Ecology 91*, 767-781.

Karsten, G.R., Drake, H.L. (1997). Denitrifying bacteria in the earthworm gastrointestinal tract and in vivo emission of nitrous oxide (N_2O) by earthworms. *Applied and Environmental Microbiology 63*, 1878-1882.

Kim I., Deurer M., Sivakumaran S., Huh K.Y., Green S.R., Clothier B.E. (2011). The impact of soil carbon management and environmental conditions on N mineralisation. *Biology and Fertility of Soils* doi 10.007/s00374-010-0526-0.

Kirkham M.B. (2011). *Elevated carbon dioxide: impacts on soil and plant water relations.* Florida, Taylor & Francis Group. 399 p.

Klaubauf, S., Inselsbacher, E., Zechmeister-Boltenstern, S., Wanek, W., Gottsberger, R., Strauss, J., Gorfer, M. (2010). Molecular diversity of fungal communities in agricultural soils from Lower Austria. *Fungal Diversity 44*, 65–75.

Kowalchuk, G.A. (2012). Bad news for soil carbon sequestration? *Science 337*, 1049-1050.

Kowalchuk, G.A., Stephen, J.R. (2001). Ammonia-oxidizing bacteria: A model for molecular microbial ecology. *Annual Review Microbiology 55*, 485–529.

Kriticos, D.J., Webber, B.L., Leriche, A., Ota, N., Macadam, I., Bathols, J., Scott, J.K. (2012). CliMond: global high-resolution historical and future scenario climate surfaces for bioclimatic modelling. *Methods in Ecology and Evolution 3*, 53-64.

Kulmatiski, A., Beard, K.H., Stevens, J.R., Cobbold, S.M. (2008). Plant-soil feedbacks: a meta-analytical review. *Ecology Letters 11*, 980-992.

Lam, S.K., Chen, D., Norton, R., Armstrong, R., Mosier, A. (2012). Nitrogen dynamics in grain crop and legume pasture systems under elevated atmospheric carbon dioxide concentration: A meta-analysis. *Global Change Biology 18*, 2853 -2859.

Ledgard, S. (2001). Nitrogen cycling in low input legume-based agriculture, with emphasis on legume/grass pastures. *Plant and Soil 228*, 43-59.

Lukac, M., Calfapietra, C., Godbold, D.L. (2003). Production, turnover and mycorrhizal colonization of root systems of three Populus species grown under elevated CO_2 (POPFACE). *Global Change Biology 9*, 838–848.

Luke C.M., Cox P.M. (2011). Soil carbon and climate change: from the Jenkinson effect to the compost-bomb instability. *European Journal of Soil Science 62*, 5–12.

Luo, Y., Su, B., Currie, W.S., Dukes, J.S., Finzi, A., Hartwig, U., Hungate, B., McMurtrie, R.E., Oren, F., Parton, W.J., Pataki, D.E., Shaw, M.R., Zak, D.R., Field, C.B. (2004). Progressive nitrogen limitation of ecosystem responses to rising atmospheric carbon dioxide. *Bioscience 54*, 731-739.

Luscher, A., Hartwig, U.A., Suter, D., Nosberget, J. (2000). Direct evidence that symbiotic N_2 fixation in fertile grassland is an important trait for strong response of plants to elevated CO_2. *Global Change Biology 6*, 655-662.

Lynch, M.D., Thorn, R.G. (2006). Diversity of basidiomycetes in michigan agricultural soils. *Applied and Environmental Microbiology 72*, 7050-7056.

Macdonald, C.A., Thomas, N., Robinson, L., Tate, K.R., Ross, D.J., Dando, J., Singh, B.K. (2009). Physiological, biochemical and molecular responses of the soil microbial community after afforestation of pastures with Pinus radiata. *Soil Biology and Biochemistry 41*, 1642-1651.

Maestre, F.T., Castillo-Monroy, A.P., Bowker, M.A., Ochoa-Hueso, R. (2012). Species richness effects on ecosystem multifunctionality depend on evenness, composition and spatial pattern. *Journal of Ecology 100*, 317-330.

Maier, M., Schack-Kirchner, H., Hildebrand, E.E., Holst, J. (2010). Pore-space CO_2 dynamics in a deep, well-aerated soil. *European Journal of Soil Science 61*, 877-887.

Malchair, S., De Boeck, H.J., Lemmens, C.M.H.M., Merckx, R., Nijs, R., Ceulemans, R., Carnol, M. (2010). Do climate warming and plant species richness affect potential nitrification, basal respiration and ammonia-oxidizing bacteria in experimental grasslands? *Soil Biology and Biochemistry 42*, 1944-1951.

Malcolm, G.M., López-Gutiérrez, J.C., Koide, R.T., Eissenstat, D.M. (2008). Acclimation to temperature and temperature sensitivity of metabolism by ectomycorrhizal fungi. *Global Change Biology 14*, 1169–1180.

Manderscheid, R., Bender, J., Schenk, U., Weigel, H.J. (1997). Response of biomass and nitrogen yield of white clover to radiation and atmospheric CO_2 concentration. *Environmental and Experimental Botany 38*, 131-143.

Marilley, L., Hartwig, U.A., Aragno, M. (1999). Influence of an elevated atmospheric CO_2 content on soil and rhizosphere bacterial communities beneath Lolium perenee and Trifolium repens under field conditions. *Microbial Ecology 38*, 39-49.

Maron, J.L., Marler, M., Klironomos, J.N., Cleveland, C.C. (2011). Soil fungal pathogens and the relationship between plant diversity and productivity. *Ecology Letters 14*, 36-41.

Mendes, R., Kruijt, M., de Bruijn, I., Dekkers, E., van der Voort, M., Schneider, J.H.M., Piceno, Y.M., DeSantis, T.Z., Andersen, G.L., Bakker, P.A.H.M., Raaijmakers, J.M. (2011). Deciphering the rhizosphere microbiome for disease-suppressive bacteria. *Science 332*, 1097-1100.

Menge, D.N.L.and Field, C.B. (1997). Simulated global changes alter phosphorus demand in annual grassland. *Global Change Biology* 13, 2582-2591.

Mercer, C.F., Bell, N.L., Yeates, G.W. (2008). Plant-parasitic nematodes on pasture in New Zealand. *Australasian Plant Pathology 37*, 279-288.

Mercer, C.F., Grant, J.L. (1993). Reproduction of *Meloidogyne hapla* and *Heterodera trifolii* from several sites in New Zealand on resistant and susceptible lines of white clover. *Nematologica 39*, 312-321.

Mercer, C.F., Starr, J.L., Miller, K.J. (1997). Host-parasite relationships of Meloidogyne trifoliophila isolates from New Zealand. *Journal of Nematology 29*, 55-64.

Mertens, J., Broos, K., Wakelin, S.A., Kowalchuk, G.A., Springael, D., Smolders, E. (2009). Bacteria, not archaea, restore nitrification in a zinc-contaminated soil. *The ISME Journal 3*, 916-923.

Milcu, A., Paul, S., Lukac, M. (2011). Belowground interactive effects of elevated CO_2, plant diversity and earthworms in grassland microcosms. *Basic and Applied Ecology 12*, 600-608.

Mohan, J.E., Clark, J.S., Schlesinger, W.H. (2007). Long-term CO_2 enrichment of a forest ecosystem: Implications for forest regeneration and succession. *Ecological Applications 17*(4), 1198-1212.

Montealegre, C.M., van Kessel, C., Russelle, M.P., Sadowsky, M.J. (2002). Changes in microbial activity and composition in a pasture ecosystem exposed to elevated atmospheric carbon dioxide. *Plant and Soil 243*, 197-207.

Müller K., Deurer M., Newton P.C.D. (2010a). Is there a link between elevated atmospheric carbon dioxide concentration, soil water repellency and soil carbon mineralization? *Agriculture, Ecosystems and Environment 139*, 98-109.

Müller K., Deurer M., Slay M., Aslam T., Carter J.A., Clothier B.E. (2010b). Environmental and economic consequences of soil water repellency under pasture. *Proceedings New Zealand Grasslands Association 72*, 207–210.

Newton, P.C.D., Allard, V., Carran, R.A., Lieffering, M. (2006). Impacts of elevated CO_2 on a grassland grazed by sheep: the New Zealand FACE experiment. In: Nosberger, J., Long, S.P., Norby, R.J., Stitt, M., Hendrey, G.R., Blum, H. (Eds.), *Managed Ecosystems and CO_2: Case Studies, Processes, and Perspectives* (pp. 157-171). Berlin, Germany: Springer-Verlag.

Newton, P.C.D., Clark, H., Bell, C.C., Glasgow, E.M. (1996). Interaction of soil moisture and elevated CO_2 on the above-ground growth rate, root length density and gas exchange of turves from temperate pasture. *Journal of Experimental Botany 47*, 771-779.

Newton, P.C.D., Clark, H., Bell, C.C., Glasgow, E.M., Tate, K.R., Ross, D.J., Yeates, G.W., Saggar, S. (1995). Plant growth and soil processes in temperate grassland communities at elevated CO_2. *Journal of Biogeography 22*, 235-240.

Newton, P.C.D., Lieffering, M., Bowatte, W.M.S.D., Brock, S.C., Hunt, C.L., Theobald, P.W., Ross, D.J. (2010). The rate of progression and stability of progressive nitrogen limitation at elevated atmospheric CO_2 in a grazed grassland over 11 years of Free Air CO_2 enrichment. *Plant and Soil 336*, 433-441.

Nielsen, U.N., Ayres, E., Wall, D.H., Bardgett, R.D. (2011). Soil biodiversity and carbon cycling: a review and synthesis of studies examining diversity-function relationships. *European Journal of Soil Science 62*, 105-116.

Niklaus, P.A., Alphei, J., Ebersberger, D., Kampichler, C., Kandeler, E., Tscherko, D. (2003). Six years of in situ CO_2 enrichment evoke changes in soil structure and soil biota of nutrient-poor grassland. *Global Change Biology 9*, 585-600.

Niklaus, P.A., Kandeler, E., Leadley, P.W., Schmid, B., Tscherko, D., Körner, C. (2001). A link between plant diversity, elevated CO_2 and soil nitrate. *Oecologia 127*, 540-548.

Orwin, K.H., Buckland, S.M., Johnson, D., Turner, B.L., Smart, S., Oakley, S., Bardgett, R.D. (2010). Linkages of plant traits to soil properties and the functioning of temperate grassland. *Journal of Ecology 98*, 1074-1083.

Orwin, K.H., Kirschbaum, M.U.F., St John, M.G., Dickie, I.A. (2011). Organic nutrient uptake by mycorrhizal fungi enhances ecosystem carbon storage: a model-based assessment. *Ecology Letters 14*, 493-502.

Orwin, K.H., Wardle, D. A. (2005). Plant species composition effects on belowground properties and the resistance and resilience of the soil microflora to a drying disturbance. *Plant and Soil 278*, 205-221.

Parfitt, R.L. (2009). Allophane and imogolite, role in soil biogeochemical processes. *Clay Minerals 44*, 125-145.

Parfitt, R.L., Giltrap, D.J., Whitton, J.S. (1995). Contribution of organic matter and clay minerals to the cation exchange capacity of soil. *Communications in Soil Science and Plant Analysis 26*(9-10), 1343-1355.

Parfitt, R.L., Percival, H.J., Dahlgren, R.A., Hill, L.F. (1997). Soil and soil solution chemistry under pasture and radiata pine in NZ. *Plant and Soil 91*, 279-290.

Parfitt, R.L., Schipper, L.A., Baisden, W.T. (2006). Nitrogen inputs and outputs for New Zealand in 2001 at national and regional scales. *Biogeochemistry 80*, 71-88.

Parsons, A., Rowarth, J., Thornley, J., Newton, P. (2011). Primary production of grasslands, herbage accumulation and use, and impacts of climate change. In: Lemaire, G., Hodgson, J., Chabbi, A. (eds.) Grassland productivity and ecosystem services. CABI, Pp 3-18.

Patterson, D.T., Westbrook, J.K., Joyce, R.J.V., Lingren, P.D., Rogasik, J. (1999). Weeds, insects, and diseases. *Climatic Change 43*, 711-727.

Pautasso, M., Döring, T.F., Garbelotto, M., Pellis, L., Jeger, M.J. (2012). Impacts of climate change on plant diseases—opinions and trends. *European Journal of Plant Pathology 133*, 295-313.

Pester, M., Bittner, N., Deevong, P., Wagner, M., Loy, A. (2010). A ‘rare biosphere’ microorganism contributes to sulfate reduction in a peatland. *The ISME Journal 4*, 1591-1602.

Pfender, W.F., Vollmer, S.S. (1999). Freezing temperature effect on survival of Puccinia graminis subsp. graminicola in Festuca arundinacea and Lolium perenne. *Plant disease 83*, 1058-1062.

Pritchard, S.G. (2011). Soil organisms and global climate change. *Plant Pathology 60*, 82-99.

Quested, H., Eriksson, O., Fortunel, C., Garnier, E. (2007). Plant traits relate to whole-community litter quality and decomposition following land use change. *Functional Ecology 21*, 1016-1026.

Rawls W.J., Pachepsky Y.A., Ritchie J.C., Sobecki T.M., Bloodworth H., (2003). Effect of soil-organic matter on soil water retention. *Geoderma 116*, 61–76.

Reverchon, F., Xu, Z., Blumfield, T.J., Chen, C., Abdullah, K.M. (2012). Impact of global climate change and fire on the occurrence and function of understorey legumes in forest ecosystems. *Journal of Soils and Sediments 12*, 150-160.

Richter, D.D., Bacon, A.R., Mobley, M.L., Richardson, C.J., Andrews, S.S., West, L., Wills, S., Billings, S., Cambardella, C.A., Cavallaro, N., DeMeester, J.E., Franzluebbers, A.J., Grandy, A.S., Grunwald, S., Gruver, J., Hartshorn, A.S., Janzen, H., Kramer, M.G., Ladha, J.K., Lajtha, K., Liles, G.C., Markewitz, D., Megonigal, P.J., Mermut, A.R., Rasmussen, C., Robinson, D.A., Smith, P., Stiles, C.A., Tate, R.L., Thompson, A., Tugel, A.J., van Es, H., Yaalon, D., Zobeck, T.M. (2011). Human-soil relations are changing rapidly: proposals from SSSA's cross-divisional soil change working group. *Soil Science Society of America Journal 75*, 2079-2084.

Ridley, G.S., Bain, J., Bulman, L.S., Dick, M.A., Kay, M.K. (2000). Threats to New Zealand's indigenous forests from exotic pathogens and pests.

Rillig, M.C. (2005). A connection between fungal hydrophobins and soil water repellency? *Pedobiologia 49*, 395-399.

Rillig, M.C., Mummey, D.L. (2006). Mycorrhizas and soil structure. *New Phytologist 171*, 41-53.

Robinson D.A., Hockley N., Cooper D., Emmet B.A., Keith A.M., Lebron L., Emmett, Reynolds B., Tipping E., Tye A.M., Watts C.W., Whalley W.R., Black H.I.J., Warren G.P., Robinson J.S. (2012). Natural capital and ecosystem services, developing an appropriate soils framework as a basis for valuation. *Soil Biology and Biochemistry, in press*.

Robinson D.A., Jackson B.M., Clothier B.E., Dominati E.J., Marchant S.C., Cooper D.M., Bristow K.L. (2013). Advances in soil ecosystem service concepts, models and applications for Earth System Life Support. *Vadose Zone Journal, in press.*

Robinson D.A., Lebron I., Ryel R.J., Jones S.B. (2010). Soil water repellency: A method of soil moisture sequestration in pinyon-juniper woodland. *Soil Science Society of America Journal 74*, 624–634.

Robinson, D.A., Hockley, N., Cooper, D., Emmet, B.A., Keith, A.M., Lebron L., Reynolds, B., Tipping, E., Tye, A.M., Watts, C.W., Whalley, W.R., Black, H.I.J., Warren, G.P., Robinson, J.S. (2013). Natural capital and ecosystem services, developing an appropriate soils framework as a basis for valuation. *Soil Biology and Biochemistry 57*, 1023-1033.

Rogers, A., Ainsworth, E.A., Leakey, A.D.B. (2009). Will elevated carbon dioxide concentration amplify the benefits of nitrogen fixation in legumes? *Plant Physiology 151*, 1009-1016.

Ross, D.J., Newton, P.C.D., Tate, K.R. (2004). Elevated [CO_2] effects on herbage production and soil carbon and nitrogen pools and mineralization in a species-rich, grazed pasture on a seasonally dry sand. *Plant and Soil 260*, 183-196.

Ross, D.J., Newton, P.C.D., Tate, K.R., Luo, D. (2013). Impact of a low level of CO_2 enrichment on soil carbon and nitrogen pools and mineralization rates over ten years in a seasonally dry, grazed pasture. *Soil Biology and Biochemistry 58*, 265-274.

Rousk, J., Baath, E. (2011). Growth of saprotrophic fungi and bacteria in soil. *FEMS Microbiology Ecology 78*, 17-30.

Ruf, A., Kuzyakov, Y., Lopatovskaya, O. (2006). Carbon fluxes in soil food webs of increasing complexity revealed by ^{14}C labelling and ^{13}C natural abundance. *Soil Biology & Biochemistry 38*, 2390-2400.

Ruiz-Lozano, J.M., Porcel, R., Azcón, R., Aroca, R. (2012). Regulation by arbuscular mycorrhizae of the integrated physiological response to salinity in plants. New challenges in physiological and molecular studies. *Jorunal of Experimental Botany* 63, 4033–4044

Ryle, G.J.A., Powell, C.E., Davidson, I.A. (1992). Growth of white clover, dependent on N_2 fixation, in elevated CO_2 and temperature. *Annals of Botany 70*, 221-228.

Sackett, T.E., Classen, A.T., Sanders, N.J. (2010). Linking soil food web structure to above- and belowground ecosystem processes: a meta-analysis. *Oikos 119*, 1984-1992.

Salles, J.F., Poly, F., Schmid, B., Le Roux, X. (2009). Community niche predicts the functioning of denitrifying bacterial assemblages. *Ecology 90*, 3324-3332.

Saxe, H., Ellsworth, D.S., Heath, J. (1998). Tree and forest functioning in an enriched CO_2 atmosphere. *New Phytologist 139*, 395-436.

Schimel, J.P., Gulledge, J. (1998). Microbial community structure and global trace gases. *Global Change Biology 4*, 74-758.

Schimel, J.P., Schaeffer, S.M. (2012). Microbial control over carbon cycling in soil. *Frontiers in Microbiology 3*. Article 348.

Schloss, P.D., Handelsman, J. (2006). Toward a census of bacteria in soil. *PLoS Comp Biol 2*(7), e92. DOI: 10.1371/journal.pcbi.0020092

Schortemeyer, M., Hartwig, U.A., Hendrey, G.R., Sadowsky, M.J. (1996) Microbial community changes in the rhizospheres of white clover and perennial ryegrass exposed to Free Air Carbon dioxide Enrichment (FACE). *Soil Biology and Biochemistry 28*, 1717-1724.

Serraj, R., Sinclair, T.R., Allen, L.H. (1998) Soybean nodulation and N_2 fixation response to drought under carbon dioxide enrichment. *Plant, Cell and Environment 21*, 491-500.

Shaw, M.W., Osborne, T.M. (2011). Geographic distribution of plant pathogens in response to climate change. *Plant Pathology 60*, 31-43.

Sheik, C.S., Beasley, W.H., Elshahed, M.S., Zhou, X., Luo, Y., Krumholz, L.R. (2011). Effect of warming and drought on grassland microbial communities. *The ISME Journal 5*, 1692-1700.

Singh, B.K., Bardgett, R.D., Smith, P., Reay, D.S. (2010). Microorganisms and climate change: Terrestrial feedbacks and mitigation options. *Nature Reviews Microbiology 8*, 779-790.

Smits, T.H.M., Wick, L.Y., Harms, H., Keel, C. (2003). Characterization of the surface hydrophobicity of filamentous fungi. *Environmental Microbiology 5*, 85-91.

Souza, L., Belote, R.T., Kardol, P., Weltzin, J.F., Norby, R.J. (2010). CO_2 enrichment accelerates successional development of an understory plant community. *Journal of Plant Ecology 3*, 33-39.

Srivastava, D.S., Cardinale, B.J., Downing, A.L., Duffy, J.E., Jouseau, C., Sankaran, M., Wright, J. P. (2009). Diversity has stronger top-down than bottom-up effects on decomposition. *Ecology 90*, 1073-1083.

Stamou, G.P., Asikidis, M.D., Argyropoulou, M.D., Iatrou, G.D. (1995). Respiratory responses of oribatid mites to temperature changes. *Journal of Insect Physiology 41*, 229-233.

Stelzer, C.P. (2005). Evolution of rotifer life histories. *Hydrobiologia 546*, 335-346.

Stevnbak, K., Maraldo, K., Georgieva, S., Bjørnlund, L., Beier, C., Schmidt, I.K., Christensen, S. (2012) Suppression of soil decomposers and promotion of long-lived, root herbivorous nematodes by climate change. *European Journal of Soil Biology 52*, 1-7.

Sticht, C., Schrader, S., Giesemann, A., Weigel, H.J. (2006). Effects of elevated atmospheric CO_2 and N fertilization on abundance, diversity and C-isotopic signature of collembolan communities in arable soil. *Applied Soil Ecology 34*, 219-229.

Strickland, M.S., Lauber, C., Fierer, N., Bradford, M. A. (2009). Testing the functional significance of microbial community composition. *Ecology 90*, 441-451.

Sturhan, D., Wouts, W.M., Grandison, G.S., Barber, C.J. (1997). Nematode vectors of plant viruses in New Zealand. *New Zealand Journal of Zoology 24*, 309-322.

Šustr, V., Pižl, V. (2010). Temperature dependence and ontogenetic changes of metabolic rate of an endemic earthworm Dendrobaena mrazeki. *Biologia 65*, 289-293.

Sutherst, R.W., Maywald, G.F. (1985). A computerised system for matching climates in ecology. *Agriculture, Ecosystems and Environment 13*, 281-299.

Svobodová, E., Trnka, M., Dubrovský, M., Semerádová, D., Eitzinger, J., Žalud, Z., Štepánek, P. (2013). Pests occurrence model in current climate-validation study for european domain. *Acta Universitatis Agriculturae et Silviculturae Mendelianae Brunensis 61*, 204-214.

Swift, M.J., Heal, O.W., Anderson, J.M. (1979). *Decomposition in terrestrial ecosystems*. Studies in Ecology Vol 5. University of California Press, Berkley, 372 Pp.

Tarnawski, S., Hamelin, J., Jossi, M., Aragno, M., Fromin, N. (2006). Phenotypic structure of Pseudomonas populations is altered under elevated pCO_2 in the rhizosphere of perennial grasses. *Soil Biology & Biochemistry 38*, 1193-1201.

Thonon, I., Klok, C. (2007). Impact of a changed inundation regime caused by climate change and floodplain rehabilitation on population viability of earthworms in a lower River Rhine floodplain. *Science of the Total Environment 372*, 585-594.

Tilman, D., Wedin, D. (1991). Plant Traits and Resource Reduction for 5 Grasses Growing on a Nitrogen Gradient. *Ecology 72*, 685-700.

Tingey, D.T., McKane, R.B., Olszyk, D.M., Johnson, M.G., Rygiewicz, P.T., Lee, E.H. (2003). Elevated CO_2 and temperature alter nitrogen allocation in Douglas-fir. *Global Change Biology 9*(7), 1038-1050.

Torsvik, V., Ovreas, L., Thingstad, T.F. (2002). Prokaryotic diversity – magnitude, dynamics, and controlling factors. *Science 296*, 1064-1066.

Treseder, K.K., Allen, M. F. (2000). Mycorrhizal fungi have a potential role in soil carbon storage under elevated CO_2 and nitrogen deposition. *New Phytologist 147*,189-200.

Treseder, K.K., Balser, T.C., Bradford, M.A., Brodie, E.L., Dubinsky, E.A., Eviner, V.T., Hofmockel, K.S., Lennon, J.T., Levine, U.Y., MacGregor, B.J., Pett-Ridge, J., Waldrop, M.P. (2012). Integrating microbial ecology into ecosystem models: challenges and priorities. *Biogeochemistry* 109, 7-18.

Tubiello, F.N., Soussana, J.-F., Howden, S.M. (2007). Crop and pasture response to climate change. *Proceedings of the National Academy of Sciences 104*, 19686-19690

Uvarov, A.V., Kolosova, N., Kulinich, O. (2009). Combined effects of earthworms and diurnal temperature fluctuations on successional changes of litter nematode community. *Polish Journal of Ecology 57,* 685-696.

van Der Heijden, M.G.A., Ursic, K.J.N.,M., Moutoglis, P., Streitwolf Engel, R., Boller, T., Wiemken, A., Sanders, R. (1998). Mycorrhizal fungal diversity determines plant biodiversity, ecosystem variability and productivity. *Nature 396*, 69-72.

van der Krift, T.A.J., Berendse, F. (2001). The effect of plant species on soil nitrogen mineralization. *Journal of Ecolog*y *89,* 555-561.

Van der Weerden T.J., Kelliher F.M., de Klein C.A.M. (2012). Influence of pore size distribution and soil water content on nitrous oxide emissions. *Soil Research 50*,125–135.

Velbel, M.A. (1993). Temperature dependence of silicate weathering in nature: How strong a negative feedback on long-term accumulation of atmospheric CO_2 and global greenhouse warming? *Geology 21*, 1059-1062.

Verschoor, B.C., De Goede, R.G.M., De Hoop, J.W., De Vries, F.W. (2001). Seasonal dynamics and vertical distribution of plant-feeding nematode communities in grasslands. *Pedobiologia 45*, 213-233.

Wakelin, S.A., Barratt, B.I.P., Gerard, E., Gregg, A.L., Brodie, E.L., Andersen, G.L., DeSantis, T.Z., Zhou J., He, Z., Kowalchuk, G.A., O’Callaghan, M. (2013). Shifts in the phylogenetic structure and functional capacity of soil microbial communities follow alteration of native tussock grassland ecosystems. *Soil Biology & Biochemistry 57*, 675-682.

Waldrop, M.P., Balser, T.C., Firestone, M.K. (2000). Linking microbial community composition to function in a tropical soil. *Soil Biology & Biochemistry 32*, 1837.1846.

Wall, D.H., Adams, G., Parsons, A.N. (2001). *Soil Biodiversity*. In: Sala, O.E., Huber-Sannwald E., eds. *Global biodiversity in a changing environment: Scenarios for the 21st century*. Springer-Verlag, New York., Pp. 47-82.

Wang, X., Taub, D.R. (2010). Interactive effects of elevated carbon dioxide and environmental stresses on root mass fraction in plants: A meta-analytical synthesis using pairwise techniques. *Oecologi*a *163,* 1-11

Wardle, D.A. (2002). *Communities and ecosystems: linking the aboveground and belowground components.* Princeton University Press, Princeton.

Wardle, D.A., Bardgett, R. D., Klironomos, J. N., Setala, H., van der Putten, W. H., Wall, D. H. (2004). Ecological linkages between aboveground and belowground biota. *Science 304*, 1629-1633.

Watanabe, T., Bowatte, S., Newton, P.C.D. Reduced nifH gene transcript numbers indicate a reduced capacity for nitrogen fixation in nodles of white clover after exposure of grazed grassland to elevated CO_2 (in preparation).

Wertz, S., Degrange, V., Prosser, J.I, Poly, F., Commeaux, C., Freitag, T., Guillaumaud, N., Le Roux, X. (2006). Maintenance of soil functioning following erosion of microbial diversity. *Environmental Microbiology 8,* 2162-2169.

West, J.B., Hille Ris Lambers J., Lee, T.D., Hobie, S.E., Reich, P.B. (2005). Legume species identity and soil nitrogen supply determine symbiotic nitrogen-fixation responses to elevated atmospheric [CO_2]. *New Phytologist 167*, 523-530.

White, A.F., Blum, A.E. (1995). Effects of climate on chemical weathering in watersheds. *Geochimica et Cosmochimica Acta 59,* 1729-1747.

White, A.F., Blum, A.E., Bullen, T.D., Vivit, D.V., Schulz, M., Fitzpatrick, J. (1999). The effect of temperature on experimental and natural chemical weathering rates of granitoid rocks. *Geochimica et Cosmochimica Acta 63,* 3277-3291.

Wilson, M.J., Kakouli-Duarte, T. (2009). *Nematodes as environmental indicators*. Wallingford, CABI. Pp. 326.

Wolters, V., Ekschmitt, K. (1995). Earthworm effects on the use of C sources by microorganisms: Non-linear response to temperature alteration. *Biology and Fertility of Soils 19*, 109-114.

Wüst, P.K., Horn, M.A., Henderson, G., Janssen, P.H., Rehm, B.H.A., Drake, H.L. (2009). Gut-associated denitrification and in vivo emission of nitrous oxide by the earthworm families megascolecidae and lumbricidae in New Zealand. *Applied and Environmental Microbiology 75,* 3430-3436.

Yeates, G.W., Boag, B., Evans, K.A., Neilson, R. (1998). Impact of climatic changes on the distribution of Paratrichodorus minor (Nematoda: Trichodoridae) as estimated using 'CLIMEX'. *Nematologica 44,* 293-301.

Yeates, G.W., Newton, P.C.D. (2009). Long-term changes in topsoil nematode populations in grazed pasture under elevated atmospheric carbon dioxide. *Biology and Fertility of Soils 45*, 799-808.

Yeates, G.W., Tate, K.R., Newton, P.C.D. (1997). Response of the fauna of a grassland soil to doubling of atmospheric carbon dioxide concentration. *Biology and Fertility of Soils 25,* 307-315.

Young, I.A., Blanchart, E., Chenu, C., Dangerfield, M., Fragoso, C., Grimaldi, J.I., Monroszier, J. (1998). The interaction of soil biota and soil structure under global change. *Global Change Biology 4*, 703-712.

Zahid, M.I., Gurr, G.M., Nikandrow, A., Hodda, M., Fulkerson, W.J., Nicol, H.I. (2001). Survey of fungi and nematodes associated with root and stolon diseases of white clover in the subtropical dairy region of Australia. *Australian Journal of Experimental Agriculture 41*, 1133-1142.

Zanetti, S., Hartwig, U.A., Luscher, A., Hebeisen, T., Frehner, M., Fischer, M., Fischer, B.U., Hendrey, G.R., Blum, H., Nosberger, J. (1996). Stimulation of symbiotic N_2 fixation in Trifolium repens L. under elevated pCO_2 in a grassland ecosystem. *Plant Physiology 112*, 575-583.

Zhang W., Ricketts T.H., Kremen C., Carney K., Swinton S.M. (2007). Ecosystem services and dis-services to agriculture. *Ecological Economics 64*, 253–260.

4 Impacts of climate change on soil carbon cycling

Loretta Garrett[1], Roberta Gentile[2], Carolyn Hedley[3], Mike Dodd[4], Simeon Smaill[1], Miko Kirschbaum[3], Peter Beets[1], Mike Beare[2]

[1]Scion, [2]Plant and Food Research, [3]Landcare Research, [4]AgResearch

4.1 KEY MESSAGES

The changing climate will impact on the soil carbon (C) cycling processes that are strongly influenced by temperature and soil moisture content. The impacts of climate change on New Zealand's primary land based sectors will vary as a result of different land use and management. Although the soil C cycle has been well studied, there is no clear understanding of the interactions between the factors controlling soil C stocks and climate change factors i.e. carbon dioxide (CO_2), temperature and moisture. The key results are presented in Table 4.1, which shows the predicted changes to the soil C cycle process and resulting soil C stocks for cropping, pastoral (dairy, extensive grazing), and forestry. The general direction of change and level of certainty are specified.

Table 4.1: Summary of the impacts of climate change on carbon cycling and stocks.

Factors potentially affected	Cropping	Pasture intensive (e.g. dairy)	Pasture extensive (e.g. dry stock)	Forestry	Justification
Soil C supply	↗	↗	↗	↗	• Increasing temperature and CO_2 concentrations will increase Net Primary Production (NPP) where there are no soil moisture or nutrient limitations. An expected increase in soil C supply from increased root exudation and turnover. • In the absence of legumes, eCO_2 leads to plant tissue of higher C:N ratio that is less labile, resulting in reduced decomposition rates. • Extreme events reduce NPP and soil C supply, through droughts inducing moisture deficits, heavy rainfall increasing erosion risk for hill country, and increased windiness increasing erosion risk in cropping sectors.
Soil C decomposition	↗	↗	↗	↗	• Decomposition of soil organic matter is driven by soil micro-organisms and is very temperature sensitive, increasing with higher temperature across all sectors. • Seasonal changes in soil C supply are likely to alter the temperature dependence of decomposition, due to changing amounts and biochemical quality of litter. • The biochemical quality of the C supply to soils in the absence of legumes is likely to be higher, resulting in slower decomposition.
Soil C stabilisation	-	-	-	-	• Soil C stabilisation is indirectly impacted by soil C supply and soil C decomposition. An increase in soil C supply promotes increased C protection where there is no cultivation or soil disturbance. • Increased temperatures may increase or decrease soil C chemical protection. • Altered precipitation (wetting and drying cycles) may increase or decrease physical C protection.
Soil C loss - DOC	-	-	-	-	• Increase in high rainfall events has the potential to increase dissolved organic carbon (DOC) loss through drainage.
Soil C loss - erosion	↗	→	↗	↗	• Increase in windiness will increase risk of wind erosion for cropping sectors. • No anticipated change in erosion for dairy. • Increase in extreme heavy rainfall events will increase the risk of erosion for erosion-prone hill country in both the pastoral sector and forestry in the post-harvest phase.
Soil C stocks	→ ↘	→ ↘	→ ↘	→ ↘	• Increase in windiness will increase risk of wind erosion for cropping sectors. • Increase in extreme heavy rainfall events will increase the risk of erosion for erosion-prone hill country in both the pastoral sector and forestry in the post-harvest phase. • High uncertainty around the impact of CC on existing protected old C decomposition or loss across all sectors.

Direction of change and certainty in science knowledge

↗	Overall, most likely to increase	↗ ↘	Could increase, remain unchanged or decrease	■	Reasonably certain of effects
↘	Overall, most likely to decrease			▨	Neither certain nor uncertain
→	Overall, most likely to remain unchanged	-	Impacts are reasonably uncertain; therefore the direction of change cannot be predicted		

4.2 INTRODUCTION

Soil carbon (C) is an important part of the global C cycle and is critical to the soil supporting processes (e.g. nutrient cycling) and soil natural capital. The extent to which soils and soil C will play a role in climate change mitigation is not well understood. Globally, soils represent a large and extremely important C reservoir, larger than the atmospheric and vegetation C reservoirs combined (Davidson and Janssens, 2006). In New Zealand the total soil C stock to 30 cm depth is estimated at 2,890 Tg (Tate et al., 2005) and a small loss of soil C can result in a large release of respired C dioxide (CO_2) to the atmosphere. Moreover, soil organic C is linked to soil quality (Lal, 2004; 2005). There is a need to understand whether climate change will increase or decrease soil C. Although the soil C cycle has been well studied there is no clear understanding of the interactions between the factors controlling soil C stocks and climate change factors: carbon dioxide (CO_2), temperature and moisture.

Net primary production (NPP) is the initial source of C to the system, followed by additional processes whereby plant C is delivered to the soil, either on or below the soil surface (Fig. 4.1). Once part of the soil, the C may undergo decomposition and be released as CO_2 to the atmosphere, or be stabilised and protected in the short- to long-term from decomposition. Soil C may also be lost through dissolved organic carbon (DOC) transport to ground water or through erosion. Climate change is expected to impact the C cycle process with greater C inputs and faster C decomposition rates, and variable impacts on soil C stabilisation and loss.

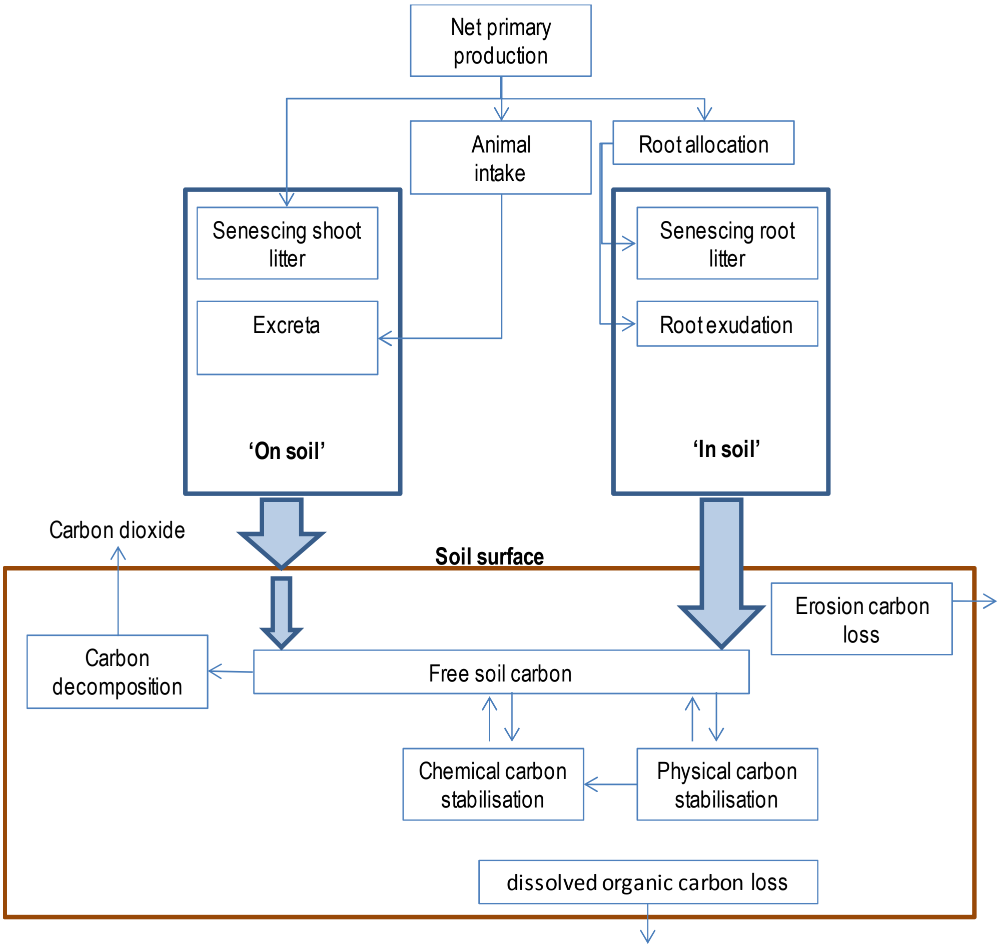

Figure 4.1: Key processes in the soil organic carbon cycle in primary production systems.

Projections made by NIWA indicate that future climate change in New Zealand will involve increases in temperature and changes in rainfall, including a greater frequency of extreme storm events and drought, as well as elevated atmospheric CO_2 concentrations. The impacts of soil erosion, a key process in soil C loss, have been reviewed (Basher et al., 2012). However, much uncertainty exists around the impacts of other key processes such as soil C inputs, decomposition and stabilisation and interactions with the N cycle on soil carbon. The changing climate is expected to have an impact on soil C stocks in the long-term, however the magnitude and direction of change in soil C at the national level is uncertain. Land use significantly impacts on soil C stocks and processes, as forests contain more biomass and capture more carbon than pastoral or cropping systems. Furthermore, site management influences the incorporation of surface C residues into the soil. It is possible that indirect effects of climate change on land use and management patterns may exceed direct effects on soil C.

This section reviews current understanding of the likely changes in soil C processes and soil C stocks under a changing climate for the three primary production systems (pastoral, arable and production forestry) and their association with the main climate change factors: elevated atmospheric CO_2 (eCO_2), increased temperature, changes in rainfall and increased frequency of extreme weather events. The interactions of eCO_2, temperature and rainfall are also considered.

We examine the impact of climate change on soil C through four key questions:

1. What are the impacts of climate change on the **supply of C to the soil** (C inputs from plant production)?
2. What are the impacts of climate change on the **decomposition rate of soil C** (plant litter and soil organic matter decomposition)?
3. What are the impacts of climate change on **soil C stabilisation and loss processes** as they affect the soil's carbon storage capacity?
4. What are the impacts of climate change on the change of **soil C stocks** under current and future farm and forestry systems?

4.3 WHAT ARE THE IMPACTS OF CLIMATE CHANGE ON THE SUPPLY OF CARBON TO THE SOIL (FROM PLANT PRODUCTION)?

Scope and limitations

Net primary production (NPP) is the initial source of C to the systems but there are additional processes whereby that plant C is delivered to the soil (either on or below the soil surface, Fig. 4.1). Other sections cover the decomposition of organic C on/in soil leading to losses of stabilised soil C and respiration. An important caveat is that few studies actually measure NPP or the processes delivering C to the soil – most measure biomass accumulation, which in crops may be harvested yield, and in pastures is actually an estimate of animal intake. So the assumption is frequently made that increases in measured biomass accumulation reflect increases in NPP.

Effect of elevated atmospheric carbon dioxide

Elevated CO_2 concentration is expected to increase NPP in all plant production systems (Friend, 2010) in the absence of other limiting factors (e.g. soil fertility). This has been demonstrated in empirical studies, both laboratory and field-based, across the three primary production systems of relevance to New Zealand: pastoral (Newton et al., 2001), arable (Morgan et al., 2005), and forestry (DeLucia et al., 2005). Leaf photosynthesis increases by 30-50% in C_3 grasses and 10-25% in C_4 grasses with a doubling of CO_2 (Soussana & Lüscher, 2007). The difference between C_4 and C_3 species is due to the greater efficiency of the Rubisco C-assimilation enzyme in C_3 plants as stomatal CO_2 increases (Sage & Kubien, 2007). Despite the observed downregulation of photosynthetic capacity, stimulation of leaf photosynthesis is maintained over time (Leakey et al., 2009) but short and long-term plant growth responses differ due to soil feedback factors affecting nutrient availability (Soussana & Lüscher, 2007; Newton et al., 2010; Norby et al., 2010). Increased NPP generally translates into increased yield; across a range of crops, a doubling of CO_2 increases biomass production by ~30% in C_3 species but <10% in C_4 species (Hatfield et al., 2011). In forests, growth has been shown to increase by a median of 23% across a range of site productivity levels (Norby et al., 2005). This general effect is supported by CO_2 enrichment experiments on young pine trees in New Zealand (Atwell et al., 2003). Recent estimates of the magnitude of the expected plant growth response are typically lower than earlier estimates. Specifically, free air CO_2 enrichment (FACE) studies have indicated on average 50% less response compared to chamber studies, due to the elimination of light and temperature artefacts associated with chamber studies (Long et al., 2006).

Effect of increased temperature

Plant species vary in the defined range of minimum-optimum-maximum temperatures within which growth occurs. Thus, increased temperature will either increase or decrease NPP and biomass accumulation in plant production systems, depending on location and species (Hatfield et al., 2011). Plants are generally able to acclimate to temperature changes, and in general increased temperature will result in increased NPP in grassland (Wan et al., 2005; Zha et al., 2013) and biomass (Liu et al., 2011; Lu et al., 2013), particularly in the temperate climates of New Zealand where damaging high temperatures are not projected and growth-limiting low temperatures will become less frequent. Increased temperatures generally accelerate phenological development, which can shorten growing seasons for annual crops and thus reduce NPP (Kirschbaum, 2004a). On the other hand increased temperatures will lengthen the growing season for perennial crops such as pastures and trees (Way & Oren, 2010). In mixed-species pasture, shifts in the dominance of species in response to temperature changes are likely, e.g. C_3 to C_4 grasses, with the latter better adapted to higher temperatures (Sage & Kubien, 2007). This will change the pattern of production response to temperature. The effects of temperature on plant production are likely to be further modified over time as higher temperatures increase nutrient supply via greater mineralisation rates (Niklaus 2007), which may mitigate expected CO_2 effects on nutrient limitation (Luo et al., 2004).

Effect of changes in rainfall

Moisture availability has a large influence on NPP through stomatal conductance and resultant photosynthetic rates (Lawlor & Cornic, 2002). Water deficits in particular will have detrimental effects on NPP and biomass accumulation in all plant production systems (Barker et al., 1985; Prince et al., 2001; Richardson et al., 2002). On the other hand, increased rainfall intensity can also reduce NPP via excessive soil moisture levels (McFarlane et al., 2003) and cause crop damage (Rosenzweig et al., 2002), though in the short-term the associated tissue senescence may represent an increase in C supply to soil. For production forests, small changes in rainfall projected for New Zealand are expected to have relatively minor effects on tree growth, most likely overshadowed by CO_2 and temperature effects (Kirschbaum et al., 2012b). For cropping systems, negative effects on yield are projected in a limited number of dryland sites (Teixeira & Brown, 2012). Pastoral production appears likely to be the most negatively impacted sector from a drought perspective, with model projections from the Ecoclimate project indicating production losses in the North Island and Eastern regions arising from increased frequency and severity of drought years (Baisden et al., 2008).

Effect of interactions between CO_2, temperature and rainfall changes

Interactions between CO_2, temperature and rainfall changes are readily observed (e.g. Shaw et al., 2002; Tubiello et al., 2007; Morgan et al., 2011) and thus very important in determining actual future NPP outcomes for any given location and species combination (Kirschbaum, 2004a).

Due to increases in photorespiration with increasing temperature in C_3 species (Sage & Kubien, 2007) the CO_2 effect on primary production is greater at higher temperatures. Decreases in stomatal conductance and canopy evapo-transpiration under eCO_2 mean that the water use efficiency (WUE) of all plants is increased (Kimball & Idso 1983; Wand et al., 1999) and reduced depletion of soil water is observed (Newton et al., 2006). Thus, the positive effect of eCO_2 will be enhanced under drier conditions for both C_3 and C_4 species due to higher WUEs (Leakey et al., 2009; Hatfield et al., 2011). So in general plants should be less sensitive to lower precipitation under eCO_2 and this may mitigate losses in NPP due to temperature-induced moisture stress. In the longer term, the increased growth of C_4 species at higher temperatures may be offset by eCO_2-induced nutrient limitation (Dodd et al., 2010).

Effect of increasing frequency of extreme weather events

There is very little information on the effects of increasing frequency of extreme weather events, because few studies have addressed these effects (Tubiello et al., 2007), with the exception of some modeling studies focused on crop yield losses (e.g. Rosenzweig et al., 2002). Strong winds are associated with lower tree growth rates (Watt et al., 2010) and increase the risk of tree damage (Dunningham et al., 2012). There are indications of lower average rates of long term crop yield increase in areas with more extreme conditions (Porter & Semenov, 2005). Extreme climatic events that involve disturbance of plant communities are likely to lead to reductions in the dominance of existing species in favour of invasive species adapted to disturbance (e.g. C_4 annual grasses compared to C_3 perennial grasses, White et al., 2001) with variable effects on NPP.

At larger scales the effects of more frequent extreme events on erosion rates in hill country will reduce long-term pasture productivity by ~20% (Rosser & Ross 2011), and also forest productivity, but to a lesser extent because of the reduced incidence of shallow landslide erosion under forests (Reid & Page, 2002). There may be as-yet unexplored interactions between the direct effects of CO_2, temperature and rainfall on plant productivity (and thus plant cover and root mass) that mitigate or exacerbate the vulnerability of soils to erosion (Basher et al., 2012).

Effects on supply of carbon to soil

Changes in NPP can be expected to translate directly into changes in overall C supply to the soil interface in terms of shoot litter, root inputs and animal excreta (in grazed pastures). However, the expected balance between these pathways may also change, with resultant impacts on the degree to which C is retained in soil. Root inputs in particular have been shown to increase under eCO_2 in pastures, in terms of both root exudation and turnover (Allard et al., 2005; Allard et al., 2006; Newton et al., 2006). This effect has also been shown in forests (Norby et al., 2004; Phillips et al., 2011). Increased root turnover in pastures in response to increased temperature was attributed to greater availability of photosynthate and nutrients (Fitter et al., 1999). In grazed pastures the balance of C supply under eCO_2 appears to have shifted toward root inputs and away from shoot and excreta inputs (Ross et al., 2013). This has implications for soil C stabilisation if, as has been suggested, most soil C is derived from root inputs (Rasse et al., 2005). Conversely, the C in root exudates is highly labile, so greater inputs of this material are unlikely to contribute to soil C stabilisation, and may act to reduce stabilisation by promoting the decomposition of more recalcitrant C (Allard et al., 2006; Hyvönen et al., 2007).

In addition, the biochemical quality of the C supply to soils is likely to alter (see section 3.6), with implications for the processes that decompose C in soils. In the absence of legumes, eCO_2 can lead to plant tissue of higher C:N ratio that is less labile (Norby & Cotrufo, 1998; Saxe et al., 1998; Körner, 2000; Niklaus et al., 2001; Newton et al., 2010) and will have reduced decomposition rates. However, changes in plant tissue quality do not always correlate to changes in litter quality, as N re-sorption during senescence can vary with the amount of N present in the plant tissue, with more N translocated from low C:N tissue compared to high C:N tissue. This effect produces litter that is essentially of equivalent quality despite originating from plant tissue of differing quality (Norby & Cotrufo, 1998), and has been observed in grazed pastures (Allard et al., 2004). Further research into this process is required to understand the relative significance for litter quality, although it is apparent it will not be significant across all sectors.

In systems where legumes are present, their greater growth responsiveness to eCO_2 (Newton et al., 2006) has been found to maintain or increase the quality of litter inputs by increasing nitrogen availability, leading to production of organic matter of lower C:N ratio (Allard et al., 2005). Moreover, in grazed pastures the presence of legumes and the effects of animal decoupling of C and N in excreta can alter C supply (Newton et al., 2006). Increased proportion of legumes in eCO_2 pastures can lead to increased litter mass loss rates (assumed to equate to C supply via decomposition), which can be mitigated by slower mass loss rates from animal excreta (Allard et al., 2004).

Changes in the combinations of plant species and edaphic conditions may lead to different rooting depth profiles and thus changes in the input of root-derived C through the soil profile. This is important because inputs deeper in soil are regarded as having slower turnover times, leading to greater stabilisation of soil C (Baisden et al., 2011). Plant root profiles will also respond to changing rainfall and temperature patterns as these impact the depth distribution of soil water content.

Changes in the attributes of soil biotic communities will also impact on soil C supply, and are reviewed in section 3.5. Changes to organic matter decomposition, which also impacts on soil C supply, are reviewed in section 4.4. The implications of changes in soil C supply for soil C storage are discussed in sections 4.5 and 4.6.

Modelling

Given the large ranges in potential responses of different plant species to climate change, and the uncertainty regarding what climatic changes will actually occur, modelling studies represent the only feasible way of making crop- and site- specific projections on the effects of climate change on NPP. This approach also eliminates the substantial cost of multi-factor empirical studies that incorporate the necessary interactions (Mikkelsen 2008). Consequently, numerous such analyses have been undertaken across many production systems globally (e.g., Riedo et al., 2000; Yang et al., 2005). The additional benefit of modelling is that specific effects on soil C supply (beyond NPP) can be assessed. The drawback is that models are open to critique about the degree to which they are validated and incorporate all the relevant underpinning processes (e.g. CO_2 effects, microbial processes) and long-term feedbacks (e.g. nutrient supply). Three recent modelling approaches for the pastoral, arable and production forest sectors in New Zealand are worth noting, since they have included CO_2, temperature and moisture effects and interactions:

- Teixeira & Brown (2012) used crop-specific ‘Plant’ modules within the process-based model APSIM to investigate the effects of predicted changes in temperature and rainfall under two emission scenarios on crop yields across four regions and two soil types. The results indicated both positive (due to CO_2 fertilisation and greater WUE) and negative (due to shorter growth cycles and reduced water availability) effects on yield which were species and site specific.
- Kirschbaum et al. (2012b) used the process-based model CenW to investigate the effects of projected changes in temperature and rainfall (from three emissions scenarios) on wood productivity across all of New Zealand at a resolution of 0.25°. In general, wood productivity was affected more by projected temperature changes than predicted rainfall changes, resulting in increased productivity in the south, but decreased productivity in the north and at lower altitudes.
- Lieffering et al. (2012) used the ‘AgPasture’ module of APSIM to investigate the effects of two emissions scenarios on the seasonal pasture production patterns of case study farms in three regions. The results showed increased overall production in the Waikato and Southland and a slight decrease in Hawke’s Bay, but all regions showed changes in seasonal patterns which were then used to examine farm system impacts and adaptations.

4.4 WHAT ARE THE IMPACTS OF CLIMATE CHANGE ON THE DECOMPOSITION RATE OF SOIL CARBON (PLANT LITTER AND SOIL ORGANIC MATTER DECOMPOSITION)?

Temperature dependence of soil organic matter decomposition

One of the key factors determining the Earth’s response to climate change is the temperature dependence of soil organic matter decomposition (SOMD) (e.g. Jones et al., 2005; Sitch et al., 2008; Kirschbaum, 2010). If global warming leads to significant losses of soil C it will constitute a dangerous feedback that could significantly increase future warming (e.g. Cox et al., 2000; Hayes et al., 2011). Despite ongoing research efforts the ranges of uncertainty predicted by modelling of the temperature response function of organic matter decomposition are large (Fig. 4.2b), and the resultant soil C changes (Fig. 4.2a) are of similar magnitude to the cumulative C release from land-use change or half of the cumulative fossil fuel emissions since the beginning of industrialisation (Denman et al., 2007; Sitch et al., 2008). These global models can be applied to New Zealand using our projected temperature changes as reported in Section 2.

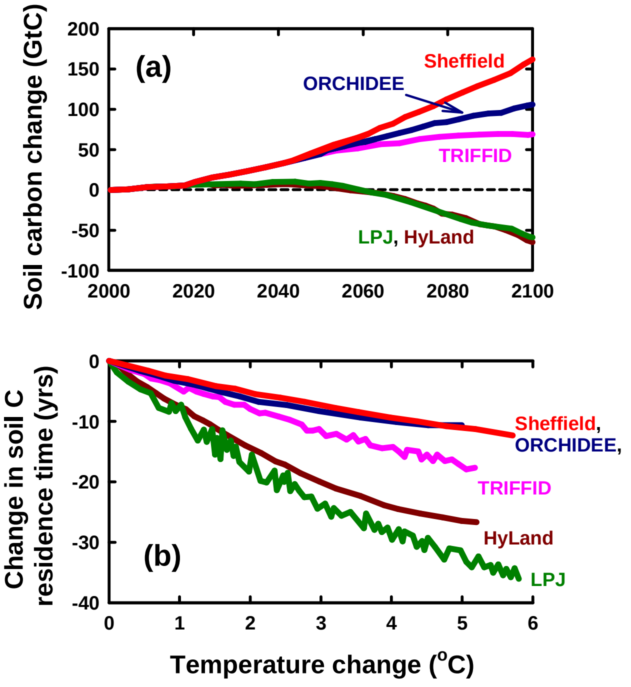

Figure 4.2: Five dynamic global vegetation models, run under SRES A1F1 scenario (a) and calculated extra-tropical mean residence time of soil C (b). Tropical residence times are similar between models. Redrawn from Sitch et al. (2008).

Therefore, different empirical approaches give different apparent sensitivities. Jones et al. (2005) showed that use of a simple one-pool organic carbon model leads to a predicted greater soil C feedback (i.e. greater effect on SOMD) compared to other models with a number of pools with different turnover times, the latter being a more accurate simulation. The various predictions from global modelling studies thus hinge on the model structure that is employed (Jones et al., 2005) and parameterisation, especially the strength of the temperature-decomposition relationship (Sitch et al., 2008; Kirschbaum, 2010). This is partly because of our incomplete understanding of SOMD, which is evolving as new methods emerge to study the nature of organic materials. Therefore SOMD mechanisms are a critical focus of current climate change research, with modelling playing an important role in our understanding.

The temperature sensitivity of soil organic matter decomposition can be described using a "Q_{10}" relationship, i.e. the proportional increase in activity per 10-degree increase in temperature. This sensitivity changes with temperature, with a proportionally greater change in activity at lower temperatures than at high temperatures (Fig. 4.3). The Q_{10} has been shown to exceed 6-fold at temperatures below 6°C, but be less than 4-fold above 10°C (Fig. 4.3).

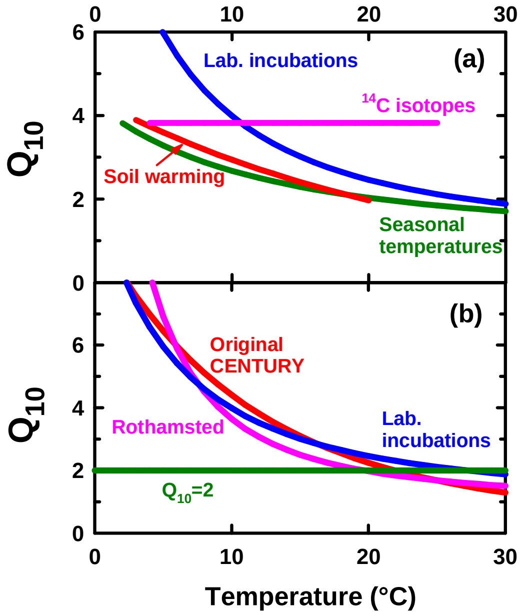

Figure 4.3: Temperature sensitivity as inferred from four different experimental approaches (a) and as used in the two most widely used soil organic matter models together with a simple Q_{10} function with Q_{10} = 2 (b). Redrawn from Kirschbaum (2000).

Temperature dependencies have been obtained through laboratory incubations (Kirschbaum, 1995, 2000), measurements of soil respiration with seasonally varying temperatures (Lloyd & Taylor, 1994), soil warming experiments and study of ^{14}C enrichment of soil organic matter along altitudinal gradients (Trumbore et al., 1996). These different methods derive reasonably consistent estimates of the temperature dependence of SOMD rates once interactions between the inherent temperature sensitivity and changing substrate availability has been explicitly taken into account (Kirschbaum, 2004b, 2013). In essence, when conditions are favourable for rapid decomposition, the resulting fast rate can deplete the pool of available substrate, leading to reduced subsequent decomposition rates. The rate of SOMD may become constrained by inadequate substrate supply to the decomposing micro-organisms. This has been noted in soil warming experiments (Kirschbaum, 2004b; Eliasson et al., 2005) where it can limit the stimulatory effect of soil warming after a number of years of experiments (e.g. Luo et al., 2001). It can also be a problem in laboratory experiments, although if recognised its extent and importance as a confounding factor can be minimised under these conditions (e.g. Nicolardot et al., 1994).

Substrate quality and quantity affects soil organic matter decomposition

Substrate quality will affect its biodegradability; for example, the seasonal variations in pasture quality and growth result in seasonal variations in the availability of labile substrate, and this seasonal variability confounds the temperature dependence of SOMD (see Section 4.2). The soil organic matter is constantly replenished through new litter influx, and the varying substrate supply of different plant species at different times of the year should be incorporated into climate change and SOMD models to improve their prediction accuracy. Therefore, changes in the pool of labile substrate typically counteract the effect of temperature in that these pools are usually smaller at times with most favourable temperatures for decomposition, i.e. summer time. It illustrates that strong intrinsic temperature dependence can easily turn into a weaker apparent temperature dependence due to varying substrate availability.

Impact of increasing temperature at cold and warm sites

These empirically-derived temperature sensitivities have also been used in the two most widely used soil organic matter models, Roth-C and CENTURY. There is also an important difference between temperature dependencies for short-term and longer-term applications. Including short-term seasonal temperature fluctuations in models, instead of annually averaged values, changes SOMD predictions. The greater the annual temperature range, the greater the deviation from the predicted response based on annual mean temperature alone. This is further illustrated in Fig. 4.4, which shows relative annual decomposition activity as a function of mean annual temperature, with the symbols representing global locations with various temperature ranges. The graph illustrates several key aspects. For the same annual mean temperature, total decomposition activity increases with magnitude of the annual temperature range. Soils with a low annual mean temperature, but a wide temperature range, experience a larger part of the year at high temperature than a soil with the same mean temperature but a lesser temperature range.

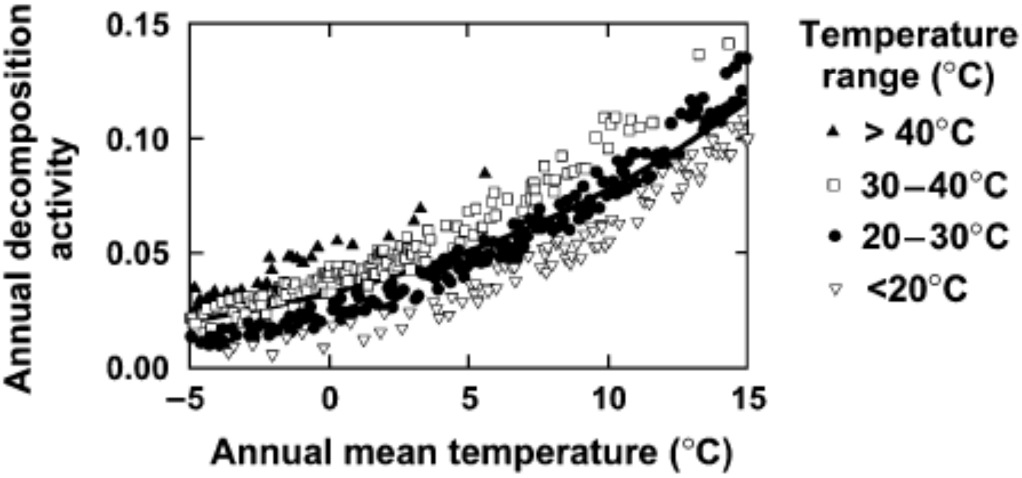

Fig. 4.4: Calculated annual decomposition activity (relative to the rate of optimum temperature) as a function of annual mean temperature. Different symbols refer to the annual temperature range at respective locations (Kirschbaum, 2010).

In colder places, there are more sites with a greater annual temperature range (e.g. at high altitudes in New Zealand). This means that the negative correlation between annual mean temperature and the temperature range partly compensates for the lower SOMD at lower temperature and the derived temperature response is less steep for annual mean temperature responses based on data from the actual distribution of temperatures across the world than would be based on a simple application of the short-term temperature dependence that ignores this interaction with seasonal temperatures (Kirschbaum, 2010). Models using short time steps (e.g. monthly) lead to reduced sensitivity of soil C stock estimates to warming, especially for cold regions, than would be predicted for simulations using annual time steps. This effect also helps to reconcile some apparent differences in temperature dependencies obtained by different workers using different approaches.

However, there are a number of complicating features of this simple picture. For example, Dungait et al. (2012) recently provided evidence that the conceptual slow and resistant soil organic carbon (SOC) pools used in computer simulation models (e.g. in RothC and CENTURY) are questionable because contemporary analytical approaches suggest that the chemical composition of these pools is not necessarily predictable from their chemical composition, i.e. that there is not necessarily a “recalcitrant” fraction. Specifically, the concept of biochemical recalcitrance, i.e. a molecular structure inherently resistant to microbial decomposition, which has previously been widely accepted, is now called into question because of the lack of an adequate molecular or mechanistic definition. Dungait et al. (2012) instead suggested that decomposition of SOM is a function of accessibility by microorganisms in the soil matrix, and not chemical stability per se.

Impact of soil microbes and soil natural capital

The physical pathway connecting the decomposer microorganism with the organic matter is the route that has to be navigated before the organic matter can be decomposed, and is influenced by motility of microorganism or the movement of exo-enzyme and the organic matter, pH, soil pore size, length, connectivity and tortuosity, strength of the soil organic matter (SOM) sorption on soil particles, occlusion between clay layers and within aggregates, efficiency of enzyme activity, as well as temperature. This research therefore suggests that SOMD is impacted by soil physical condition and biological activity as well as its chemical make-up, and models that account for movement through the soil matrix of all factors required for decomposition (i.e. microbes, organic material (e.g. root growth), moisture, temperature, nutrients) may give greater insights into the factors that control SOMD and how they may respond to future global warming.

Modelling

Conceptual models are used to understand how the soil ecosystem impacts on SOMD, and Conant et al. (2011) presented a model that distinguishes SOM available for decomposition from that which can be assimilated into microbial biomass. Their model distinguishes the different steps of the decomposition process (uptake mechanisms and microbial catabolism) from the processes that make SOM available for decomposition (adsorption/desorption and aggregate turnover). They conclude that understanding the net effect of rising temperature on soils requires understanding of all these component processes and their interactions, because decomposability of the bulk soil OM is a function of several independent processes (Kleber, 2010).

Apparent differences in model predictions can also be resolved by considering co-varying factors, such as the size of the substrate pools. Most global models still do not use nutrient cycle feedback limitations even though it is likely that the inclusion of nutrient cycling would lessen any response to external driving factors. When conditions are favourable for rapid decomposition, then the fast rate can deplete the pool of available substrate leading to reduced subsequent decomposition rates (Kirschbaum, 2013). Therefore changing nutrient substrate supply confounds the effect of temperature on SOMD.

However the more critical question for system responses to global warming is the relative temperature dependence of decomposition of the more recalcitrant fractions of organic matter. There is active debate going on as to whether fractions with different recalcitrance have the same or different temperature dependencies, with a theory being advanced that increasing recalcitrance of organic matter should be correlated with greater temperature dependence (e.g. Bosatta & Agren, 1999). The debate is in part fuelled by the emerging new definition of “recalcitrance” as being physically protected by the soil’s structural matrix (Dungait et al., 2012), rather than chemically inert to decomposition.

Water limitation impacts

Another climate change impact on SOMD is water limitation, because drier soils inhibit decomposition processes (Moyano et al., 2012). Suseela et al. (2012) examined how warming and altered precipitation affected the rate and temperature sensitivity of heterotrophic respiration (R_h). Drought reduced R_h both annually and during the growing season. During the summer, when R_h was highest, they found evidence for a threshold, hysteretic response to soil moisture: R_h decreased rapidly when volumetric soil moisture dropped below ~15% or exceeded ~26%, but R_h increased gradually when soil moisture rose from the lower threshold. It is possible that this is explained by these threshold values representing the optimum soil moisture range for microbial activity and plant growth in their study's loamy over gravelly sandy loam soil. This is an important interaction, not only because climate change may cause changes in water availability through changed precipitation patterns, but also because warming itself will lead to enhanced rates of water loss and thus drier soils unless warming is also accompanied by increased rates of precipitation. The inhibitory effects of drying of soils will partly negate the warming-enhanced stimulation of SOMD, but this interaction has not yet been explored quantitatively, and may be offset once drought has broken by decomposition of the organic carbon that has been stabilised and mineralised during the dry period.

Summary

This technical review section has discussed climate change impacts on soil organic matter decomposition, and the uncertainty associated with existing models. Ecosystem responses to climate change can be non-additive and nonlinear (Burkett et al., 2005) and both plants and microbes can exhibit threshold responses. If we can identify these nonlinearities and thresholds we will be better positioned to reduce some of the uncertainties associated with our climate change decision making (Zhou et al., 2008).

4.5 WHAT ARE THE IMPACTS OF CLIMATE CHANGE ON SOIL CARBON STABILISATION AND LOSS PROCESSES AS THEY AFFECT THE SOIL'S CARBON STORAGE CAPACITY?

Soil C stabilisation processes do not protect C indefinitely, but rather serve to reduce the rate of C decomposition relative to unprotected C (Baldock & Skjemstad, 2000). The main mechanisms of soil C stabilisation that reduce C availability for decomposition include physical protection within soil aggregates and chemical protection through organo-mineral associations (Baldock & Skjemstad, 2000; Six et al., 2002a; von Lützow et al., 2006). Physical protection involves the occlusion of soil organic C within soil aggregates where it is physically isolated from decomposers and conditions are less suitable for aerobic decomposition (Tisdall & Oades, 1982). Chemical protection refers to binding interactions of soil organic C with mineral surfaces of silt and clay particles or metal oxides (Hassink, 1997). In addition to mineralisation to CO_2, soil C may also be lost from a system due to DOC leaching to surface or groundwater or erosion.

We have little understanding of the specific processes involved in soil C stabilisation and DOC fluxes, let alone how these might respond to changing climate variables. Greater elevated atmospheric CO_2 concentrations will have indirect impacts on soil processes by increasing primary productivity and C supply (section 4.2). Increased temperatures will stimulate biological activity and change chemical reactions. Changes in precipitation patterns will influence soil processes by controlling soil moisture conditions and water flows. Here we review available literature on how each of these predicted climate change effects may impact on soil C stabilisation and loss processes.

Physical protection

Soil aggregates are the units in which C is physically protected in the soil by separating it from decomposing microorganisms and creating conditions within the aggregate interior that may limit aerobic decomposition (Ladd et al., 1993; Six et al., 2004; Tisdall and Oades, 1982). Therefore, aggregate stability and turnover can play an important role in controlling C stabilisation. Major factors that influence soil aggregation include soil fauna, soil microorganisms, roots, inorganic binding agents and environmental variables (Six et al., 2004). Soil mineralogy will also affect the processes involved in aggregate formation and stabilisation, because moderately weathered soils with 2:1 clay minerals predominantly form aggregates due to organic binding agents, whereas highly weathered soils with 1:1 clay minerals and oxides may form aggregates due to electrostatic interactions (Six et al., 2002b). New Zealand contains a range of soil orders with differing mineralogy and metal oxide contents (Percival et al., 2000), which will likely influence aggregate response to climate change a on a soil specific basis. While this review explores possible direct impacts of climate change on soil aggregate stability and turnover through changing environmental variables, climate change may indirectly influence physical soil C protection due to impacts on other aggregate formation factors.

Increased atmospheric CO_2 concentrations will not directly alter soil aggregate dynamics, but will have indirect impacts due to potential increases in soil C supply and changes in C quality. Aggregate formation and degradation are dynamic processes linked to C inputs and microbial activity, thus soil structure and soil C interact to increase soil aggregation and physically protect soil C. Elevated CO_2 experiments have shown increases in aggregate-protected C when soil C stocks have increased in forest and grassland systems (Hoosbeek & Scarascia-Mugnozza, 2009; Jastrow et al., 2005). Six et al. (2001) observed an increase in aggregation and aggregate-protected C under perennial ryegrass after 6 years of elevated CO_2 even though total soil C stocks were not changed. However, Moran & Jastrow (2010) found that management effects superseded any increase in soil C or aggregate fractions due to elevated CO_2 in an arable cropping system. While physical C stabilisation should increase with elevated CO_2, this affect will be modified by the extent of change to soil C inputs and management factors.

Little information is available in the literature on temperature effects on aggregate dynamics and C protection. We have a low certainty about the impact of increased temperatures on aggregate turnover and physical C protection. Increased temperatures may both stimulate C inputs that stabilise aggregates and increase the decomposition of organic binding agents. In a prairie soil exposed to a continuous 2°C warming for 9 years, Cheng et al. (2011) did not observe any effect on soil aggregate distribution or stability. In this experiment, increased inputs in C with warming treatment were offset by accelerated decomposition resulting in no change in soil C. Thus the temperature sensitivity of aggregate stability will likely depend on the net balance between C inputs and decomposition. Freeze-thaw cycles generally increase aggregate turnover, though the effect of this disturbance depends on soil moisture, C concentration, and clay content (Six et al., 2004). An increase in temperature and decrease in frosts may reduce freeze-thaw cycles in some areas of New Zealand and have a small effect on promoting physical C protection.

Changes in future precipitation patterns will alter soil moisture content and drying and wetting cycles. Drying and wetting can be disruptive or enhancing forces and the severity or speed of each process will control the balance of aggregate turnover. Rapid drying and wetting can decrease soil aggregation due to non-uniform shrinking or swelling of the clay fraction or compression of entrapped air causing stresses and cracks to develop (Dexter, 1991). Additionally, the physical impact of raindrops may cause aggregate breakdown depending on soil cover and moisture levels (Six et al., 2004). Alternatively, slow drying can draw C into small pores and increase molecular associations between soil C and clay minerals as water films retract, thereby stabilising soil aggregates (Degens, 1997).

Soil C content may affect the stabilising influence of drying, as Haynes & Swift (1990) found that drying increased aggregate stability of high C pasture soil, but decreased aggregate stability of low C arable soil. Additionally, Park et al. (2007) observed that additions of glucose-C maintained aggregate stability during drying and wetting cycles. Similarly, drying and wetting cycles in the presence of growing plants have shown an increase in aggregate stability, which may be attributed to the presence of C sources and root exudates during drying (Materechera et al., 1994; Reid & Goss, 1982).

Aggregate disruption due to fast drying and wetting may be a short-term effect, as aggregates have been observed to become resistant to breakdown upon repeated cycles (Denef et al., 2001). This may be due to aggregate stability increasing with time or the disintegration and reorganisation of particles to form stronger bonds (Denef et al., 2001; Dexter, 1991; Materechera et al., 1994). The impact of climate change on aggregate stability and protection of soil C will depend on the changes in rainfall distribution. Possibly areas in which rainfall decreases during the wet winter months or areas in which rainfall increases during the dry summer months will experience greater wetting and drying cycles. These physical stresses may increase or decrease the physical protection of soil C in aggregates depending on factors such as the rate of moisture change, C availability and soil mineralogy. Thus we have a low to moderate certainty about the direction of change in physical C stabilisation due to changing precipitation patterns.

Chemical protection

Soil carbon may be stabilised from decomposition by chemical or physicochemical interactions with soil minerals and metal oxides (Six et al., 2002a; von Lützow et al., 2006). Several studies have shown soil C content to increase with increasing clay content due to the greater reactive surface area of clay particles increasing the soil chemical stabilisation capacity (Hassink, 1997; Six et al., 2002a). However, in an analysis of soil C contents of New Zealand pasture soils, Percival et al. (2000) found that pyrophosphate-extractable Al was better correlated with C than clay content, suggesting that C complexation with Al-oxides may be a dominant mechanism of C stabilisation in these soils. Chemical stabilisation of soil C potentially involves multiple bonding mechanisms between mineral surfaces and C compounds including ligand exchange, cation bridging, electrostatic interaction, hydrophobic interactions, hydrogen bonding, and van der Waals forces (Sollins et al., 1996; von Lützow et al., 2006). The type of bonding present in a given soil will depend on the nature of the soil C compounds as well as the soil minerals present, and will likely involve multiple mechanisms. Adsorption and desorption processes in soils are not well understood due to this heterogeneity and complexity of bonding interactions. Hence we have great uncertainty in establishing any trends of climate change impacts on the chemical protection of soil C until we can understand the mechanisms involved.

As with soil C physical protection, elevated atmospheric CO_2 concentrations may have indirect impacts on C chemical protection by increasing the quantity of C inputs. The stabilisation of C by clay minerals would be expected to increase with increased soil C, but there are few studies examining this specific stabilisation mechanism. Increases have been found in silt and clay-associated C in response to elevated CO_2 for whole soil (Jastrow et al., 2000) and aggregate-associated fractions (Hoosbeek & Scarascia-Mugnozza, 2009).

Thus increases in soil C in response to elevated CO_2 should increase the association of C with soil minerals and metal oxides.

As soil C adsorption and desorption are chemical processes, they should be subject to kinetic theory and their rates influenced by temperature. However, due to the complexity of the interactions involved, we have little information about the activation energies of these processes (Davidson & Janssens, 2006). Conant et al. (2011) recently reviewed the literature on temperature controls of adsorption reactions. They divided C-mineral interactions into high-affinity and low-affinity reactions. High-affinity reactions are regulated by thermodynamics with adsorption as an exothermic process and desorption as an endothermic process. Increased temperatures should favour desorption over adsorption and result in decreased stabilisation of C for high-affinity reactions. Conversely, low-affinity reactions are slower and determined by the diffusion of C compounds to mineral surfaces. Thus increased temperatures should enhance the diffusion of C substrates and also promote binding sites on minerals due to desorption of high-affinity reactions. Therefore, increased temperatures should result in increased C stabilisation for low-affinity reactions. These opposing predictions for the temperature dependence of adsorption processes highlight the need to understand the binding mechanisms involved in soil C protection in order to adequately predict the impact of increased temperatures on C stabilisation.

Changes in future precipitation patterns will alter soil moisture contents and mineral weathering. According to adsorption isotherm theory, the quantity of C adsorbed will increase with increasing C concentrations in solution (Stevenson, 1994). Concentrations of DOC increase upon rewetting (Kalbitz et al., 2000), so C adsorption would be expected to increase after dry periods. Conversely, high precipitation and soil moisture contents can dilute DOC concentrations (Kalbitz et al., 2000) and thus favour desorption processes reducing C protection. Berhe et al. (2012) conducted a field experiment in a Californian grassland simulating increased future rainfall during winter and spring seasons. They found that the seasonality of changes in precipitation influenced changes in soil C stabilisation, with increased rainfall during the wet winter season decreasing soil C stabilisation by Fe and Al oxides. Conversely, increased precipitation should increase mineral weathering rates and increase allophane and extractable Al, thereby promoting C stabilisation (Percival et al., 2000).

Dissolved organic carbon transport

Dissolved organic C transport is a potential pathway of soil C loss if it is leached through the soil profile into groundwater reserves or enters surface water flow. In New Zealand, DOC exports to streams in Waikato catchments under pasture and forest have been measured at 20.2-27.4 kg ha^{-1} yr^{-1} (Quinn & Stroud, 2002). Concentrations and fluxes of DOC in the soil are controlled by biotic and abiotic processes involving the nature of the organic matter, microbial activity, soil mineralology, soil solution properties, and environmental conditions (Kalbitz et al., 2000). Ghani et al. (2007) measured DOC concentrations by chemical extraction in New Zealand surface soils and found large variability and seasonal fluctuations in concentrations ranging from 73-718 mg C kg^{-1} soil. The greatest concentrations were found in allophanic soils in Waikato and during the winter season. Climate change may modify DOC concentrations and fluxes from the soil profile by altering environmental variables that drive flux processes.

Elevated atmospheric CO_2 may indirectly impact DOC concentrations and fluxes by altering soil C inputs and soil moisture regimes. Any increase in C inputs due to CO_2

stimulation of plant productivity is likely to increase soil C pools including DOC. In the New Zealand FACE experiment, Ross et al. (2013) observed that extractable C concentrations increased after a 10 year exposure to elevated CO_2. In a German FACE experiment involving a cropping system, Siemens et al. (2012) found that elevated CO_2 increased DOC leaching not due to higher DOC concentrations but because of reduced evapotranspiration leading to increased soil moisture content and groundwater recharge. Increased soil moisture with elevated CO_2 treatment was also observed in the New Zealand FACE experiment (Ross et al., 2013). Thus increased CO_2 levels may indirectly increase the potential for DOC leaching due to increased DOC concentrations and increased water flow through the soil profile.

Dissolved organic C flux is the balance of processes that both release and remove carbon from solution. Increased temperature both stimulates soil organic mineralisation and the release of DOC as well as microbial decomposition and removal of DOC leading to no clear trend in DOC flux (Kalbitz et al., 2000). Examination of DOC fluxes in forest soils has shown temperature to have a positive effect (Borken et al., 2011) and no effect (Michalzik et al., 2001) on DOC concentrations. A meta-analysis by Lu et al. (2013) found that DOC pools increased by 12.1% with experimental warming. Therefore, increases in temperature may lead to increased DOC losses.

Altered precipitation is the climate change factor that should have the greatest impact on potential DOC losses. Hydrological controls are generally more important for DOC fluxes than biotic factors (Kalbitz et al., 2000). Concentrations of DOC increase upon rewetting of dried soil and under anaerobic conditions (Kalbitz et al., 2000). Dissolved organic C fluxes increase with increased precipitation and water movement through the soil profile (Borken et al., 2011; Kalbitz et al., 2000; Michalzik et al., 2001). These observations indicate that DOC losses will increase in areas receiving increased precipitation, but decrease in the areas where precipitation decreases. An increase in heavy precipitation events and flood risk with climate change will also increase potential DOC losses. Water movement and solute leaching through the soil profile will be influenced by hydrological conditions and soil type (e.g. Ghani et al. (2010) observed higher DOC losses in a gley soil than allophanic soil under pastures in New Zealand).

Erosion

Erosion can negatively impact on soil C through disturbance and removal. Climate change impacts on erosion in New Zealand have been recently reviewed by Basher et al. (2012). They concluded that increased incidence of storm rainfalls would increase shallow landsliding, earthflows, gully and sheet erosion, which may be counteracted in the north and east due to increased temperatures and lower rainfalls. Increased drought periods and increased windiness would also increase risk of wind erosion, particularly in the east of the country. Many areas in the east of both islands with highest potential for erosion (landslides, gully erosion and earthflows) are projected to have a decrease in mean annual rainfall, so the impact of climate change will depend on changes to extreme rainfall and extra-tropical cyclone activity. The precipitation extremes projected would possibly have the greatest impact on soil loss, particularly as the 100-year return period for 24 hour extreme storms is projected to double. Soil C moved off site may become buried lower in the catchment or eventually in ocean sinks offsetting this loss (Dymond, 2010).

4.6 WHAT ARE THE IMPACTS OF CLIMATE CHANGE ON SOIL CARBON STOCKS UNDER CURRENT AND FUTURE FARM AND FORESTRY SYSTEMS?

Climate change impacts on soil carbon stocks

Globally, soils represent a large and extremely important C reservoir, larger than the atmospheric and vegetation C reservoirs combined (Kirschbaum, 2000). Soil C stocks are determined by the balance between C inputs and outputs, which will vary with each soil-climate-land use/management combination. Soil organic C stocks can increase or decrease when the soil C cycle processes controlling the input-output balance are changed. Global warming is likely to reduce soil organic C by stimulating rates of decomposition, while increased CO_2 levels will simultaneously increase soil organic C supply through enhanced net primary production. How much C is stabilised in the soil is influenced by the change in soil C supply and decomposition, as well as soil type. Global warming is likely to reduce stabilisation and increased precipitation could remove more dissolved organic C, with heavy rainfall events increasing the risk of soil C loss through erosion. Here we review available literature on how each of these predicted climate change effects may impact on New Zealand's soil C stocks.

The quantification of soil C stock change as a result of climate change is limited by the data available and an understanding of how soil C cycle processes respond to climate variables and their interactions over the long term. The impact of individual climate variables on soil C cycle processes has a moderate level of certainty, with a moderate to high level of certainty for the impact on soil C decomposition, in predicting a change in soil C stocks.

Effect of increased temperature

Temperature changes significantly impact on soil C decomposition with modelled equilibrium changes from laboratory incubation data showing that with every degree increase in temperature, soil C stocks decrease by 5–6%, with more pronounced losses from soils at lower temperatures with small seasonal temperature variation (Kirschbaum, 2000). With the slow turnover of soil organic matter pools, it is likely, however, to take many centuries before these equilibrium changes are fully realised. The long term impact of global warming, in particular on decomposition, is expected to be influenced by seasonal temperature variations changing substrate availability and substrate C quality, which could weaken the temperature dependencies measured in laboratory incubations (Kirschbaum, 2010, 2013).

Effect of elevated carbon dioxide

Results from the New Zealand FACE experiment with increasing CO_2 in the atmosphere (475 ppm) measured an increase in soil C stocks of 1.2% per year in a sandy soil under pasture (Ross et al., 2013), which is consistent with the mean value from a meta analysis of international literature by De Graaff et al. (2006). The soil C response to increased atmospheric CO_2 concentration depends on soil nitrogen availability, with soil C stocks only increasing under high nitrogen availability (De Graaff et al., 2006). These results indicate that there would be greater soil C sequestration under high nitrogen input systems in New Zealand. The projected increase in atmospheric CO_2 to values of 480–530 ppm for New Zealand by mid century would indicate that where nitrogen is not limiting we could expect an increase in soil C supply and soil C stocks under elevated CO_2 only.

Effect of change in precipitation

Climatic changes in precipitation are projected to vary over New Zealand and with season (see Table 2.1). The projected changes in precipitation will impact on soil moisture availability, of which limited moisture negatively impacts on soil C decomposition and resulting soil C stocks (Kirschbaum, 2010). Where rainfall is plentiful, precipitation changes could have both positive and negative impacts. A New Zealand study that re-sampled pastoral soil after 20-30 years (Schipper et al., 2007) did find that decadal scale differences in seasonal patterns of temperature and rainfall (warmer and dryer) were associated, although not significantly, with the decadal scale patters in soil C and N dynamics (reduced soil C stocks under warmer and dryer conditions).

Impact of soil type

The different soil types in New Zealand have significant differences in C stocks. This is due to the influence of different soil properties on soil C stabilisation, and climate factors that influence soil processes (Dodd et al., 2011). An analysis of the upper ranges of mineral soil C stocks in New Zealand found Organic soils were the highest, followed by Podzol and Allophanic soils. The lowest soil C stocks were found in weakly developed soils such as Recent and Raw soils (Jones et al., 2012).

Under a changing climate the susceptibility of the C stored in different soil types may be influenced by how the soil C is protected. For example, soil types with low chemical activity (e.g. Raw soils) could potentially lose a higher percentage of soil C through accelerated decomposition under a changing climate than soils in which the majority of soil C is chemically protected (e.g. Podzol and Allophanic soils). For soil types that have a low soil C stock, losing even a low percentage can negatively impact on the soil quality and the ability of the soil to provide ecosystem services (Knoepp et al., 2000; Lal et al., 2007). Conversely, even a small percentage change in C stocks for soil types that have large C stocks could result in a large C release into the atmosphere, resulting in a possible critically important feedback effect for future atmospheric CO_2 concentrations (Kirschbaum, 2010; Baldock et al., 2012).

New Zealand’s soils can have protected soil C that has come from historical indigenous forest land cover (Beets et al., 2002; Meder et al., 2007). This soil C (old carbon) can be protected by the mineral soil for a considerable length of time, buffering any change; for example, the spatial variability in historical soil C can be greater than current land-use change effects. The amount of historical protected C in different soil types is also variable and uncertain, but for some soils the bulk of the soil C is from indigenous forest cover. We know how climate change effects impact on newly added soil C, but are uncertain about whether old protected soil C is less vulnerable to climate change. Given that historical C can dominate the total stock in many New Zealand soils, the effect of climate change on the historic C fraction should be of considerable interest, because if this is liberated the potential for change in C stocks is large. Limited understanding of the mechanisms protecting historical soil C means that uncertainties remain in predicting the soil C stock response to climate change scenarios for New Zealand.

Impact of land management

The impact of climate change on soil C stock is confounded by type of land management practice. Under a changing climate, the land management practices that will have the most impact on the soil C cycle and resulting soil C stocks will be soil disturbance resulting in vulnerability to erosion and loss (Kirschbaum et al., 2009). Soil disturbance can often go beyond 30 cm depth (Yang & Wander, 1999; Oliver et al., 2004), the depth to which soil C stocks are currently reported (Tate et al., 2005). Not only does erosion remove soil C from the site, but the resultant erosion scars tend to decrease productivity (20% in pastoral) relative to intact land, thereby reducing soil C supply (Rosser & Ross, 2011). This loss may be offset by soil C moved off site becoming buried lower in the catchment or eventually in ocean sinks (Dymond, 2010).

Effect of climatic extremes

Projected extreme climatic events are often localised, however, are likely to have the greatest impact on New Zealand's soil C stocks in both the short and long term:

- The projected precipitation extremes will probably have the most significant impact on soil C stocks on site by increasing erosion risk (Basher et al., 2012).
- The projected extreme temperatures resulting in more hot days will impact on the soil moisture balance, soil C cycle processes and net soil C stocks.
- Increasing evapotranspiration under a warming climate and an increase in drought magnitude and frequency will reduce soil moisture, thereby reducing NPP and the supply of C to the soil. It can be expected that where soil moisture is already a limiting factor, soil C stocks will be relatively unaffected. Where soil moisture is not limiting, positive and negative impacts on soil C stocks are expected.
- Extreme wind events are also expected to increase the risks of NPP reduction (Dunningham et al., 2012) and wind erosion (Basher et al., 2012), both of which could have a negative impact on soil C stocks.
- Finally, fire risk is projected to increase. Some charcoal can remain in the soil in a stable form, resulting in an increase in soil C stocks (von Lützow et al., 2006). Fire negatively impacts on NPP and soil C supply to the soil.

Total soil carbon stocks

Limited information about New Zealand soil C exists, so predicting the magnitude of change on total C stocks by modelling of the effects of a changing climate on New Zealand's total soil C have a low level of certainty. Although the soil C cycle has been well studied there is no clear understanding of the interactions between the factors controlling soil C stocks and climate change. Predictions made to dates indicate that:

- Modelled soil C stock changes due to climate change scenarios for New Zealand show that the net change in soil C stocks, excluding erosion, is expected to be small (Kirschbaum, 2000) or decrease (Kirschbaum et al., 2012b) over the coming centuries.
- Modelled results for planted forests predict an average C loss of 1.5% with increasing CO_2 concentrations, with lesser reductions from regions with higher growth enhancements that partly offset the faster C decomposition rate under climate change (Kirschbaum et al., 2012b).
- There are no modelled predictions on soil C stock change specifically made for other New Zealand sectors.
- A review from a European perspective concluded that there is little evidence for an overall combined positive or negative impact of climate change on soil C stocks, and that any impact is expected to be far less than the indirect effects of land use change, land use and land management impacts (Schils et al., 2008). This is also expected to be the case in New Zealand.

There are two types of uncertainty related to impacts on soil and soil C stocks: one is due to incomplete knowledge, and the other to the diversity of possible combinations of factors in the real world. The degree of certainty of the underlying science is presented in Table 4.1. Given the large range of potential responses of soil C stocks to climate change and the uncertainty regarding which climatic changes will actually occur and where, modelling studies represent the only feasible way of making site-specific projections.

Land management adaptation impacts on soil carbon stocks

Land management changes are likely with climate change (Dynes et al., 2010; Clark et al., 2012), and this in turn will have implications for soil C stocks. Examples are increased irrigation of pastoral and cropping systems where water limits growth, and change in species or genotypes that perform better under the changing climate. Increased use of nitrogen (N) fertiliser to enhance NPP where nitrogen becomes limiting is also likely (see nitrogen cycling section).

Climate change adaptation to alleviate moisture limitations or droughts through irrigation will increase the soil moisture status, removing moisture limitation to soil C decomposition or soil C supply. In a long term South Island irrigation study in Winchmore, Kelliher et al. (2012) found that soil C stocks to 1 m depth were 32% (9.1 t/ha irrigated vs 13.4 t/ha dryland) lower due to irrigation as required during summer. They estimated that the irrigation treatment had increased soil C supply by 36% but increased soil C losses via respiration by 97%. Long-term observations in the top 75 mm of soil in the same experiment (Schipper et al., 2012) showed that this difference in stocks was maintained despite both treatments accumulating soil C over the 60 years since border-dyke establishment. Globally, there are few published studies on irrigation effects on soil C in grasslands (e.g. Rixon, 1966; Xiao et al., 2007; Denef et al., 2008), and unique site factors are highly influential (Conant, 2001) and generalisations cannot be made.

Change in the use of different species or genotypes to adapted to climate change will likely impact on the soil C supply to the soil. Plant functional types influence C distribution in the soil profile through biomass allocation and root distribution effects on the placement and quantity of C inputs (Fornara & Tilman, 2008; Jobbágy & Jackson, 2000). Changes in crop and pasture species, particularly those involving deeper rooting species, may have some impacts on soil C stocks or on processes influencing C stocks (Skinner et al., 2006). Moreover, in planted forests the impact of different tree species on soil C stocks is uncertain. Broad forest types (e.g. broadleaf forests) are often referred to in the international literature on soil organic C instead of individual species, because the lack of data on single species results in considerable uncertainty about their impacts. Broadleaf tree species typically have higher soil organic C stocks than coniferous species (Guo & Gifford, 2002; Paul et al., 2002; Laganière et al., 2010), and N-fixing tree species sequester more soil organic C than other tree species (Resh et al., 2002; Binkley, 2005; Kasel et al., 2011). This suggests it is safe to conclude that a change in tree species in response to climate change will impact on the soil C stocks.

Increased use of nitrogen (N) fertiliser is not expected to change soil C stocks in high N input systems in New Zealand where N supply does not limit NPP. Where N is limiting, or becomes limiting through climate change induced Progressive Nitrogen Limitation (PNL), N fertiliser application will increase the soil C supply. Increased soil C stocks in response to N fertiliser applications have been observed in N-limited New Zealand planted forests due to increased soil C supply, despite elevated rates of soil C decomposition induced by the N fertiliser (Huang et al., 2011). Further potential outcomes of increased soil C supply for soil C stocks are discussed in detail in the soil C supply section.

The greatest impact of management adaptation to climate change on soil C stocks would be any change that reduces soil disturbance and vulnerability to erosion and loss. This is likely to be influenced by best management practices and practices to sequester soil C or minimise soil erosion (Paustian et al., 1997; Lal, 2005; Prescott, 2010; Basher et al., 2012). Moreover, land use change is likely to occur with climate change, particularly afforestation of erosion-prone hill country, and impacts on soil C stocks of these changes have been evaluated elsewhere (Kirschbaum et al., 2012a).

4.7 REFERENCES

Ainsworth, E.A., Davey, P.A., Hymus, G.J., Osborne, C.P., Rogers, A., Blum, H., Nösberger, J., Long, S.P. (2003). Is stimulation of leaf photosynthesis by elevated carbon dioxide concentration maintained in the long term? A test with Lolium perenne grown for 10 years at two nitrogen fertilization levels under Free Air CO_2 Enrichment (FACE). *Plant, Cell & Environment 26,* 705-714.

Allard, V., Newton, P.C.D., Lieffering, M., Soussana, J.-F., Grieu, P., Matthew, C. (2004). Elevated CO_2 effects on decomposition processes in a grazed grassland. *Global Change Biology 10,* 1553-1564.

Allard, V., Newton, P.C.D., Lieffering, M., Soussana, J.F., Carran, R.A., Matthew, C. (2005). Increased Quantity and Quality of Coarse Soil Organic Matter Fraction at Elevated CO_2 in a Grazed Grassland are a Consequence of Enhanced Root Growth Rate and Turnover. *Plant and Soil 276,* 49-60.

Allard, V., Robin, C., Newton, P.C.D., Lieffering, M., Soussana, J.F. (2006). Short and long-term effects of elevated CO_2 on Lolium perenne rhizodeposition and its consequences on soil organic matter turnover and plant N yield. *Soil Biology and Biochemistry 38,* 1178-1187.

Atwell, B.J., Henery, M.L., Whitehead, D. (2003). Sapwood development in Pinus radiata trees grown for three years at ambient and elevated carbon dioxide partial pressures. *Tree Physiology 23,* 13-21.

Baisden, W.T., Parfitt, R.L., Ross, C., Schipper, L.A., Canessa, S. (2011). Evaluating 50 years of time-series soil radiocarbon data: towards routine calculation of robust C residence times. *Biogeochemistry 112,* 129-137.

Baldock, J.A., Skjemstad, J.O. (2000). Role of the soil matrix and minerals in protecting natural organic materials against biological attack. *Organic Geochemistry 31,* 697-710.

Baldock, J.A., Wheeler, I., McKenzie, N., McBrateny, A. (2012). Soils and climate change: potential impacts on carbon stocks and greenhouse gas emissions, and future research for Australian agriculture. *Crop and Pasture Science 63,* 269-283.

Barker, D.J., Chu, A.C.P., Korte, C.J. (1985). Some effects of spring defoliation and drought on perennial ryegrass swards. *Proceedings of the New Zealand Grassland Association 46,* 57–63.

Basher, L., Douglas, G., Elliott, S., Hughes, A., Jones, H., McIvor, I., Page, M., Rosser, B., Tait, A. (2012). *Impacts of climate change on erosion processes and plant-based erosion control.* Report prepared for the Ministry of Primary Industries, May 2012.

Beets, P.N., Oliver, G.R., Clinton, P.W. (2002). Soil carbon protection in podocarp/hardwood forest, and effects of conversion to pasture and exotic pine forest. *Environmental Pollution 116,* S63-S73.

Berhe, A.A., Suttle, K.B., Burton, S.D., Banfield, J.F. (2012). Contingency in the direction and mechanics of soil organic matter responses to increased rainfall. *Plant and Soil 358,* 371-383.

Binkley, D. (2005). How Nitrogen-fixing trees change soil carbon. In: Binkley, D., Menyailo, O. (Eds.), *Tree Species Effects on Soils: Implications for Global Change*. NATO Science Series, Kluwer Academic Publishers, Dordrecht.

Borken, W., Ahrens, B., Schulz, C., Zimmermann, L. (2011). Site-to-site variability and temporal trends of DOC concentrations and fluxes in temperate forest soils. *Global Change Biology 17*, 2428-2443.

Bosatta, E., Angren, G.I. (1999) Soil organic matter quality interpreted thermodynamically. *Soil Biology and Biochemistry 31,* 1888-1891.

Burkett, V.R., Wilcox, D.A., Stottlemeyer, R., Barrow, W., Fagre, D., Baron, J., Price, L., Nielsen, J.L., Allen, C.D., Peterson, D.L., Ruggerone, G., and Doyle, T. (2005) Nonlinear dynamics in ecosystem responses to climatic change: case studies and policy implications. *Ecological Complexity 2*, 237-394.

Cheng, X., Luo, Y., Xu, X., Sherry, R., Zhang, Q. (2011). Soil organic matter dynamics in a North America tallgrass prairie after 9 yr of experimental warming. *Biogeosciences 8*, 1487-1498.

Clark, A.J., Nottage, R.A.C., Wilcocks, L., Lee, J.M., Burke, C., Kalaugher, E., Roche, J., Beukes, P., Lieffering, M., Newton, P.C.D., Li, F.Y., Vibart, R., Teixeira, E.I., Brown, H.E., Fletcher, A.L., Hernandez-Ramirez, G., Soltani, A., Viljanen-Rollinson, S., Horrocks, A., Johnstone, P., Clothier, B., Hall, A., Green, S., Dunningham, A., Kirschbaum, M.U.F., Meason, D., Payn, T., Collins, D.B.G., Woods, R.A., Rouse, H., Duncan, M., Snelder, T., Cowie, B. (2012). Impacts of Climate Change on Land-based Sectors and Adaptation Options. In, Clark, A.J.; Nottage, R.A.C.; Hansford, D. (eds) *Stakeholder Report to the Sustainable Land Management and Climate Change Adaptation Technical Working Group, Ministry for Primary Industries*, 76 p.

Conant, R.T., K. Paustian, and E.T. Elliott. (2001). Grassland management and conversion into grassland: Effects on soil carbon. *Ecological Applications 11*, 343–355.

Conant, R.T., Ryan, M.G., Ågren, G.I., Birge, H.E., Davidson, E.A., Eliasson, P.E., Evans, S.E., Frey, S.D., Giardina, C.P., Hopkins, F.M., Hyvönen, R., Kirschbaum, M.U.F., Lavallee, J.M., Leifeld, J., Parton, W.J., Steinweg, J.M., Wallenstein, M.D., Wetterstedt, J.Å.M, Bradford, M.A. (2011). Temperature and soil organic matter decomposition rates – synthesis of current knowledge and a way forward. *Global Change Biology 17*, 3392-3404.

Cox, P.M., Betts, R.A., Jones, C.D., Spall, S.A., Totterdell, I.J. (2000). Acceleration of global warming due to carbon cycle feedbacks in a coupled climate model. *Nature 408*, 184-187.

Davidson, E.A., Janssens, I.A. (2006). Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. *Nature 440*, 165-173.

Degens, B.P. (1997). Macro-aggregation of soils by biological bonding and binding mechanisms and the factors affecting these: a review. *Australian Journal of Soil Research 35*, 431-459.

Denef, K., C.E. Stewart, J. Brenner, and K. Paustian. (2008). Does long-term centerpivot irrigation increase soil carbon stocks in semi-arid agro-ecosystems? *Geoderma 145*, 121–129.

Denef, K., Six, J., Bossuyt, H., Frey, S.D., Elliott, E.T., Merckx, R., Paustian, K. (2001). Influence of dry-wet cycles on the interrelationship between aggregate, particulate organic matter, and microbial community dynamics. *Soil Biology & Biochemistry 33*, 1599.

Denman, K.L., Brasseur, G., Chidthaisong, A., Ciais, P., Cox, P.M., Dickinson, R.E., Hauglustaine, D., Heinze, C., Holland, E., Jacob, D., Lohmann, U., Ramachandran, S., da Silva Dias, P.L., Wofsy, S.C., Zhang, X. (2007). *Couplings between changes in the climate system and biogeochemistry*. In: Solomon S, Qin D et al. (Eds) Climate Change 2007: The physical science basis. Contribution of the Working Group 1 to the Fourth Assessment Report on the IPCC on Climate Change. Cambridge University Press, pp 499-587.

De Graaff, M. A., Van Groenigen, K. J., Six, J., Hungate, B., Van Kessel, C. (2006). Interactions between plant growth and soil nutrient cycling under elevated CO_2: a meta-analysis. *Global Change Biology 12*, 2077-2091.

Dexter, A.R. (1991). Amelioration of soil by natural processes. *Soil & Tillage Research 20*, 87.

Dodd, M.B., Newton, P.C.D., Lieffering, M., Luo, D. (2010). The responses of three C4 grasses to elevated temperature and CO_2 in the field. *Proceedings of the New Zealand Grassland Association 7*2, 61-66.

Dodd, M., Parfitt, R., Beare, M., Sharp, J., Gentile, R., Jones, H., & McNeill, S. (2011). *Defining upper limits on soil organic carbon in New Zealand's agricultural soils.* Report prepared for The New Zealand Agricultural Greenhouse Gas Research Centre. The New Zealand Institute for Plant & Food Research Limited, November 2011.

Dungait, J.A.J., Hopkins, D.W., Gregory, A.S., Whitmore, A.P. (2012). Soil organic matter turnover is governed by accessibility not recalcitrance. *Global Change Biology 18*, 1781-1796.

Dunningham, A., Kirschbaum, M., Payn, T., Meason, D. (2012). Chapter 7. Forestry Long-term adaptation of productive forests in a changing climatic environment. In: *Impacts of climate change on land-based sectors and adaptation options.* Ministry for Primary Industries, 30 July 2012, ISBN Online: 978-0-478-38891-6.

Dymond, J.R. (2010). Soil erosion in New Zealand is a net sink of CO_2. *Earth Surface Processes and Landforms 35*, 1763-1772.

Dynes, R.A., Payn, T., Brown, H.E., Bryant, J.R., Newton, P., Snow, V.O., Lieffering, M., Wilson, D.R., Beets, P. (2010). New Zealand's land-based primary industries and climate change: Assessing adaptation through scenario-based modelling. In: RAC Nottage, DS Wratt, JF Bornman and K Jones (Eds.) *Climate Change adaptation in New Zealand. Future scenarios and some sectoral perspectives.* New Zealand Climate Change Centre, Wellington, pp 44-55.

Eliasson, P.E., McMurtrie, R.E., Pepper, D.A., Stromgren, M., Linder, S., Agren, G.I. (2005). The response of heterotrophic CO_2 flux to soil warming. *Global Change Biology 11*, 167-181.

Elyeznasni, N., Sellami, F., Pot, V., Benoit, P., Vieublé-Gonod, L., Young, I., Peth, S. (2012). Exploration of soil micromorphology to identify coarse-sized OM assemblages in X-ray CT images of undisturbed cultivated soil cores. *Geoderma 179-180*, 38-45.

Fitter, A.H., Self, G.K., Brown, T.K., Bogie, D.S., Graves, J.D., Benham, D., Ineson, P. (1999). Root production and turnover in an upland grassland subjected to artificial soil warming respond to radiation flux and nutrients, not temperature. *Oecologia 120*, 575-581.

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

Ghani, A., Dexter, M., Carran, R.A., Theobald, P.W. (2007). Dissolved organic nitrogen and carbon in pastoral soils: the New Zealand experience. *European Journal of Soil Science 5*8, 832-843.

Ghani, A., Muller, K., Dodd, M., Mackay, A. (2010). Dissolved organic matter leaching in some contrasting New Zealand pasture soils. *European Journal of Soil Science 61*, 525-538.

Guo, L.B., Gifford, R.M. (2002). Soil carbon stocks and land use change: A meta analysis. *Global Change Biology 8*, 345-360.

Hatfield, J.L., Boote, K.J., Kimball, B.A., Ziska, L.H., Izaurralde, R.C., Ort, D., Thomson, A.M., Wolfe, D. (2011). Climate Impacts on Agriculture: Implications for Crop Production. *Agronomy Journal*, 103, 351-370.

Hassink, J. (1997). The capacity of soils to preserve organic C and N by their association with clay and silt particles. *Plant and Soil 191*, 77-87.

Hayes, D.J., McGuire, A.D., Kicklighter, D.W., Gurney, K.R., Burnside, T.J., Melilli, J.M., (2011). Is the northern high-latitude land-based CO_2 sink weakening? *Global Biogeochemical Cycles 25*, Article No.: GB3018.

Haynes, R.J., Swift, R.S. (1990). Stability of soil aggregates in relation to organic constituents and soil water content. *Journal of Soil Science 41*, 73-83.

Hoosbeek, M.R., Scarascia-Mugnozza, G.E. (2009). Increased litter build up and soil organic matter stabilization in a poplar plantation after 6 years of atmospheric CO_2 enrichment (FACE): final results of POP-EuroFACE compared to other forest FACE experiments. *Ecosystems 12*, 220-239.

Huang Z, Clinton P, Baisden T, Davis M. 2011a. Long-term nitrogen additions increased surface soil carbon concentration in a forest plantation despite elevated decomposition. *Soil Biology and Biochemistry, 43*, 302-307.

Hyvönen, R., Ågren, G.I., Linder, S., Persson, T., Cotrufo, M. F., Ekblad, A., Freeman, M., Grelle, A., Janssens, I.A., Jarvis, P.G., Kellomäki, S., Lindroth, A., Loustau, D., Lundmark, T., Norby, R.J., Oren, R., Pilegaard, K., Ryan, M.G., Sigurdsson, B.D., Strömgren, M., Van Oijen, M., Wallin, G. (2007). The likely impact of elevated [CO_2], nitrogen deposition, increased temperature and management on carbon sequestration in temperate and boreal forest ecosystems: a literature review. *New Phytologist 173*, 463-480.

Jastrow, J.D., Miller, R.M., Matamala, R., Norby, R.J., Boutton, T.W., Rice, C.W., Owensby, C.E. (2005). Elevated atmospheric carbon dioxide increases soil carbon. *Global Change Biology 11*, 2057-2064.

Jastrow, J.D., Miller, R.M., Owensby, C.E. (2000). Long-term effects of elevated atmospheric CO_2 on below-ground biomass and transformations to soil organic matter in grassland. *Plant and Soil 224*, 85-97.

Jobbágy E.G., Jackson R.B. (2000). The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications 10,* 423-436.

Jenkinson, D.S., Adams, D.E., Wild, A. (1991). Model estimates of CO_2 emissions from soil in response to global warming. *Nature 351*, 304-306.

Jones, C., McConnell, C., Coleman, K., Cox, P., Falloon, P., Jenkinson, D., Powlson, D. (2005). Global climate change and soil carbon stocks: predictions from two contrasting models for the turnover of organic carbon in soil. *Global Change Biology*: *11*, 154–166.

Jones, H., McNeill, S., Beare, M., Dodd, M., Parfitt, R., Sharp, J. (2012). *The upper range of measured C stocks in New Zealand soils and key environmental explanatory variables.* A report prepared for The New Zealand Agricultural Greenhouse Gas Research Centre. The New Zealand Institute for Plant & Food Research Limited, April 2012.

Jones, H., McNeill, S., Beare, M., Dodd, M., Parfitt, R., & Sharp, J. (2012). *The upper range of measured C stocks in New Zealand soils and key environmental explanatory variables.* A report prepared for The New Zealand Agricultural Greenhouse Gas Research Centre. The New Zealand Institute for Plant & Food Research Limited, April 2012.

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

Kasel, S., Singh, S., Sanders, G.J., Bennett, L.T. (2011). Species-specific effects of native trees on soil organic carbon in biodiverse plantings across north-central Victoria, Australia. *Geoderma 161,* 95-106.

Kelliher, F. M., Condron, L. M., Cook, F. J., Black, A. (2012). Sixty years of seasonal irrigation affects carbon storage in soils beneath pasture grazed by sheep. *Agriculture, Ecosystems & Environment 148,* 29-36.

Kimball, B.A., Idso, S.B. (1983). Increasing atmospheric CO_2: effects on crop yield, water use and climate. *Agricultural Water Management 7,* 55-72.

Kirschbaum, M.U.F. (1995). The temperature dependence of soil organic matter decomposition and the effect of global warming on soil carbon storage. *Soil Biology and Biochemistry 27,* 753-760.

Kirschbaum, M., Trotter, C., Wakelin, S., Baisden, T., Curtin, D., Dymond, J., Ghani, A., Jones, H., Deurer, M., Arnold, G., Beets, P., Davis, M., Hedley, C., Peltzer, D., Ross, C., Schipper, L., Sutherland, A., Wang, H., Beare, M., Clothier, B., Mason, N., Ward, M. (2009). *Carbon stocks and changes in New Zealand's soils and forests, and implications of post-2012 accounting options for land-based emissions offsets and mitigation opportunities*. Chapter 1. In, Landcare Research Contract Report: LC0708/174, prepared for MAF, Wellington, April 2009.

Kirschbaum, M. (2000). Will changes in soil organic carbon act as a positive or negative feedback on global warming? *Biogeochemistry 48*, 21-51.

Kirschbaum, M.U.F. (2004a). Direct and Indirect Climate Change Effects on Photosynthesis and Transpiration. *Plant Biology 6*, 242-253.

Kirschbaum, M.U.F. (2004b). Soil respiration under prolonged soil warming: are rate reductions caused by acclimation or substrate loss? *Global Change Biology 10*, 1870-1877.

Kirschbaum, M.U.F. (2006). The temperature dependence of organic-matter decomposition – still a topic of debate. *Soil Biology & Biochemistry 38*, 2510-2518.

Kirschbaum, M.U.F. (2010). The temperature dependence of organic matter decomposition: seasonal temperature variations turn a sharp short-term temperature response into a more moderate annually averaged response. *Global Change Biology 16*, 2117-2129.

Kirschbaum, M.U.F., Saggar, S., Tate, K.R., Giltrap, D.L., Ausseil, A.-G.E., Greenhalgh, S., Whitehead, D., (2012a). Comprehensive evaluation of the climate-change implications of shifting land use between forest and grassland: New Zealand as a case study. *Agriculture, Ecosystems and Environment 150,* 123-138.

Kirschbaum, M.U.F., Watt, M.S., Tait, A., Ausseil, A.-G.E. (2012b). Future wood productivity of Pinus radiata in New Zealand under expected climatic changes. *Global Change Biology 18,* 1342-1356.

Kirschbaum, M.U.F. (2013). Seasonal variations in the availability of labile substrate confound the temperature dependence of organic matter decomposition. *Soil Biology and Biochemistry 57*, 568-576.

Kleber, M., Nico, P.S., Plante, A.F., Filley, T., Kramer, M., Swanston, C., Sollins, P. (2011). Old and stable soil organic matter is not necessarily chemically recalcitrant: implications for modelling concepts and temperature sensitivity. *Global Change Biology 17*, 1097-1107.

Knoepp, J.D., Coleman, D.C., Crossley Jr., D.A, Clark, J.S. (2000). Biological indices of soil quality: An ecosystem case study of their use. *Forest Ecology and Management 138,* 357-368.

Körner, C. (2000). Biosphere responses to CO_2 enrichment. *Ecological Applications 10*, 1590-1619.

Kutsch, W.L., Bahn, M., Heinemeyer, A. (2012). *Soil Carobn Dynamics – An Integrated Methodology*. Cambridge University Press. 288 pp.

Ladd, J.N., Foster, R.C., Skjemstad, J.O. (1993). Soil structure: carbon and nitrogen metabolism. *Geoderma 56*, 401.

Laganière, J., Angers, D.A., Parè, D. (2010). Carbon accumulation in agricultural soils after afforestation: a meta-analysis. *Global Change Biology 16,* 439-453.

Lal, R., Follett, R.F., Stewart, B.A., Kimble, J.M. (2007). Soil carbon sequestration to mitigate climate change and advance food security. *Soil Science 172*, 943-956.

Lawlor, D.W., Cornic, G. (2002). Photosynthetic carbon assimilation and associated metabolism in relation to water deficits in higher plants. *Plant, Cell & Environment 25*, 275-294.

Leakey, A.D.B., Ainsworth, E.A., Bernacchi, C.J., Rogers, A., Long, S.P., Ort, D.R. (2009). Elevated CO_2 effects on plant carbon, nitrogen, and water relations: six important lessons from FACE. *Journal of Experimental Botany 60,* 2859-2876.

Lieffering, M., Newton, P.C.D., Li, F., Vibart, R. (2012). Hill country sheep and beef: impacts and adaptations to climate change. Pp. 149-187 In: *Impacts of climate change on land-based sectors and adaptation options.* Clark, A.; Nottage, R. eds. Ministry for Primary Industries.

Liu, Y., Reich, P.B., Li, G., Sun, S. (2011). Shifting phenology and abundance under experimental warming alters trophic relationships and plant reproductive capacity. *Ecology 92*, 1201-1207.

Lloyd, J., Taylor, J.A. (1994). On the temperature dependence of soil respiration. *Functional Ecology 8*, 315-323.

Long, S.P., Ainsworth, E.A., Leakey, A.D.B., Nösberger, J., Ort, D.R. (2006). Food for Thought: Lower-Than-Expected Crop Yield Stimulation with Rising CO_2 Concentrations. *Science 312*, 1918-1921.

Lu, M., Zhou, X., Yang, Q., Li, H., Luo, Y. Fang, C. Chen, J. Yang, X., Li, B., 2013. Responses of ecosystem carbon cycle to experimental warming: a meta-analysis. *Ecology 94*, 726-738.

Luo, Y., Wan, S., Hui D., Wallace, L.L. (2001). Acclimatization of soil respiration to warming in tall grass prairie. *Nature 413*, 622–625.

Luo, Y., Su, B., Currie, W.S., Dukes, J.S., Finzi, A., Hartwig, U., Hungate, B., McMurtrie, R.E., Oren, R., Parton, W.J., Pataki, D.E., Shaw, M.R., Zak, D.R., Field, C.B. (2004). Progressive nitrogen limitation of ecosystem responses to rising atmospheric carbon dioxide. *BioScience 54,* 731-739.

Materechera, S.A., Kirby, J.M., Alston, A.M., Dexter, A.R. (1994). Modification of soil aggregation by watering regime and roots growing through beds of large aggregates. *Plant and Soil* 160, 57-66.

McFarlane, N.M., Ciavarella, T.A., Smith, K.F. (2003). The effects of waterlogging on growth, photosynthesis and biomass allocation in perennial ryegrass (Lolium perenne L.) genotypes with contrasting root development. *The Journal of Agricultural Science 141,* 241-248.

Meder, R., Beets, P.N., Oliver, G.R. (2007). Multivariate analysis of IR, NIR, and NMR spectra of soil samples from different land use conversions: native forest, pasture, and plantation forest. *New Zealand Journal of Forestry Science 37*, 289-305.

Michalzik, B., Kalbitz, K., Park, J.-H., Solinger, S., Matzner, E. (2001). Fluxes and concentrations of dissolved organic carbon and nitrogen - a synthesis for temperate forests. *Biogeochemistry 52*, 173-205.

Mikkelsen, T.N. (2008). Experimental design of multifactor climate change experiments with elevated CO_2, warming and drought: the CLIMAITE project. *Functional Ecology 22*, 185-195.

Moran, K.K., Jastrow, J.D. (2010). Elevated carbon dioxide does not offset loss of soil carbon from a corn-soybean agroecosystem. *Environmental Pollution 158*, 1088-1094.

Morgan, J.A., LeCain, D.R., Pendall, E., Blumenthal, D.M., Kimball, B.A., Carrillo, Y., Williams, D.G., Heisler-White, J., Dijkstra, F.A., West, M. (2011). C4 grasses prosper as carbon dioxide eliminates desiccation in warmed semi-arid grassland. *Nature 476*, 202-205.

Moyano. F.E., Vasilyeva, N., Bouckaert, L., Cook, F., Craine, J., Yuste, J.C., Don, A., Apron, D., Formanek, P., Franzluebbers, A., Ilstedt, U., et al. (2012). The moisture response of soil heterotrophic respiration: interaction with soil properties. *Biogeosciences 9*, 1173-1182.

Newton, P.C.D., Allard, V., Carran, R.A., Lieffering, M. (2006). Impacts of elevated CO_2 on a grassland grazed by sheep: the New Zealand FACE experiment. Pp. 157-171, In: *Managed Ecosystems and CO_2. Case Studies, Processes and Perspectives*. Nosberger, J.; Long, S. P.; Norby, R. J.; Stitt, M.; Hendrey, G. R.; Blum, H. eds. Berlin, Germany, Springer-Verlag.

Newton, P.C.D., Carran, R.A., Edwards, G.R., Niklaus, P.A. (2007). Introduction. Pp. 1-8. In: *Agroecosystems in a changing climate.* Newton, P.C.D., Carran, R.A., Edwards, G.R., Niklaus, P.A. eds. Boca Raton, Taylor & Francis.

Newton, P.D., Lieffering, M., Bowatte, W.M.S., Brock, S., Hunt, C., Theobald, P., Ross, D. (2010). The rate of progression and stability of progressive nitrogen limitation at elevated atmospheric CO_2 in a grazed grassland over 11 years of Free Air CO_2 enrichment. *Plant and Soil 336*, 433-441.

Nicolardot, B., Fauvet, G., Chenby, D. (1994). Carbon and nitrogen cycling through soil microbial biomass at various temperatures. *Soil Biology and Biochemistry 26*, 253-261.

Niklaus, P.A. (2007). Climate change effects on biogeochemical cycles, nutrients and water supply. Pp. 11-52. In: *Agroecosystems in a changing climate.* Newton, P.C.D., Carran, R.A., Edwards, G.R., Niklaus, P.A. eds. Boca Raton, Taylor & Francis.

Niklaus, P.A., Glöckler, E., Siegwolf, R., Körner, C. (2001). Carbon allocation in calcareous grassland under elevated CO_2: a combined 13C pulse-labelling/soil physical fractionation study. *Functional Ecology 15*, 43-50.

Norby, R.J., Cotrufo, M.F. (1998). A question of litter quality. *Nature 396*, 17-18

Norby, R.J., DeLucia, E.H., Gielen, B., Calfapietra, C., Giardina, C.P., King, J.S., Ledford, J., McCarthy, H.R., Moore, D.J.P., Ceulemans, R., De Angelis, P.C., Finzi, A.C., Karnosky, D.F., Kubiske, M.E., Lukac, M., Pregitzer, K.S., Scarascia-Mugnozza, G.E., Schlesinger, W.H., Oren, R. (2005). Forest response to elevated CO_2 is conserved across a broad range of productivity. *Proceedings of the National Academy of Sciences of the United States of America 102*, 18052-18056.

Norby, R.J., Ledford, J., Reilly, C.D., Miller, N.E., O'Neill, E.G. (2004). Fine-root production dominates response of a deciduous forest to atmospheric CO_2 enrichment. *Proceedings of the National Academy of Sciences of the United States of America 101*, 9689-9693.

Norby, R.J., Warren, J.M., Iversen, C.M., Medlyn, B.E., McMurtrie, R.E. (2010). CO_2 enhancement of forest productivity constrained by limited nitrogen availability. *Proceedings of the National Academy of Sciences of the United States of America 107*, 19368-19373.

Oliver, G.R., Beets, P.N., Garrett, L.G., Pearce, S.H., Kimberly, M.O., Ford-Robertson, J.B., 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*, 283-295.

Park, E. J., Sul, W.J., Smucker, A.J.M. (2007). Glucose additions to aggregates subjected to drying/wetting cycles promote carbon sequestration and aggregate stability. *Soil Biology & Biochemistry 39*, 2758-2768.

Parton, W.J., Schimel, D.S., Cole, C.V., & Ojima, D.S. (1987). Analysis of factors controlling soil organic matter levels in Great Plains grasslands. *Soil Science Society of America Journal 51*, 1173-1179.

Paul, K.I., Polglase, P.J., Nyakuengama, J.G., Khanna, P.K. (2002). Change in soil carbon following afforestation. *Forest Ecology and Management 168*, 241-257.

Paustian K., Andrén O., Janzen H.H., Lal R., Smith P., Tian G., Tiessen H., Van Noordwijk M., Woomer P.L. (1997) Agricultural soils as a sink to mitigate CO_2 emissions. *Soil Use and Management 13*, 230-244.

Percival, H.J., Parfitt, R.L., Scott, N.A. (2000). Factors controlling soil carbon levels in New Zealand grasslands: Is clay content important? *Soil Science Society of America Journal 64*, 1623-1630.

Phillips, R.P., Finzi, A.C., Bernhardt, E.S. (2011). Enhanced root exudation induces microbial feedbacks to N cycling in a pine forest under long-term CO_2 fumigation. *Ecology Letters 14*, 187-194.

Porter, J.R., Semenov, M.A. (2005). Crop responses to climatic variation. *Philosophical Transactions of the Royal Society B: Biological Sciences 360*, 2121-2035.

Post, W.M., Emanuael, W.R., Zinke, P.J., Stangenberger, A.G. (1982). Soil carbon pools and world life zones. *Nature 298*, 156-159.

Prescott, C. (2010). Litter decomposition: what controls it and how can we alter it to sequester more carbon in forest soils? *Biogeochemistry 101*, 133-149.

Quinn, J.M., Stroud, M.J. (2002). Water quality and sediment and nutrient export from New Zealand hill-land catchments of contrasting land use. *New Zealand Journal of Marine and Freshwater Research 36*, 409-429.

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

Reid, J.B., Goss, M.J. (1982). Interactions between soil drying due to plant water use and decreases in aggregate stability caused by maize roots. *Journal of Soil Science 33*, 47-53.

Reid, L.M., Page, M.J. (2002). Magintude and frequency of landsliding in a large New Zealand catchment. *Geomorphology 49*, 71-88.

Resh, S.C., Binkley, D., Parrotta, J.A. (2002). Greater soil carbon sequestration under nitrogen-fixing trees compared with Eucalyptus species. *Ecosystems 5*, 217-231.

Richardson, B., Whitehead, D., McCracken, I.J. (2002). Root-zone water storage and growth of Pinus Radiata in the presence of a broom understorey. *New Zealand Journal of Forestry Science 32*, 208-220.

Rixon, A.J. (1966). Soil fertility changes in a red-brown earth under irrigated pastures. *Australian Journal of Agricultural Research 17*, 317–325.

Rosenzweig, C., Tubiello, F.N., Goldberg, R., Mills, E., Bloomfield, J. (2002). Increased crop damage in the US from excess precipitation under climate change. *Global Environmental Change 1*2, 197-202.

Ross, D.J., Newton, P.C.D., Tate, K.R., Luo, D. (2013). Impact of a low level of CO_2 enrichment on soil carbon and nitrogen pools and mineralization rates over ten years in a seasonally dry, grazed pasture. *Soil Biology and Biochemistry 58*, 265-274.

Ross, D.J., Tate, K.R., Newton, P.C.D., Clark, H. (2002). Decomposability of C3 and C4 grass litter sampled under different concentrations of atmospheric carbon dioxide at a natural CO_2 spring. *Plant and Soil 240*, 275-286.

Rosser, B.J., Ross, C.W. (2011). Recovery of pasture production and soil properties on soil slip scars in erodible siltstone hill country, Wairarapa, New Zealand. *New Zealand Journal of Agricultural Research 54*, 23-44.

Sage, R.F., Kubien, D.S. (2007). The temperature response of C3 and C4 photosynthesis. *Plant, Cell & Environment 30*, 1086-1106.

Saxe, H., Ellsworth, D.S., Heath, J. (1998) Tree and forest functioning in an enriched CO_2 atmosphere. *New Phytologist 139*(3), 395-436.

Schils, R., Kuikman, P., Liski, J., Oijen, M.V., Smith, P., Webb, J., Alm, J., Somogyi, Z., Akker, J.V.D., Billett, M., Emmett, B., Evans, C., Lindner, M., Palosuo, T., Bellamy, P., Alm, J., Jandl, R., Hiederer, R. (2008). *Review of existing information on the interrelations between soil and climate change.* (SoilClim). Final report. In, Brussels, European Communities, 2008, p. 209.

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.

Schipper, L.A., Dodd, M.B., Pronger, J., Mudge, P.L., Upsdell, M., Moss, R.A. (2012). Decadal Changes in Soil Carbon and Nitrogen under a Range of Irrigation and Phosphorus Fertilizer Treatments. *Soil Science Society of America Journal 77*, 246–256.

Siemens, J., Pacholski, A., Heiduk, K., Giesemann, A., Schulte, U., Dechow, R., Kaupenjohann, M., Weigel, H.J. (2012). Elevated air carbon dioxide concentrations increase dissolved carbon leaching from a cropland soil. *Biogeochemistry 108*, 135-148.

Shaw, M.R., Zavaleta, E.S., Chiariello, N.R., Cleland, E.E., Mooney, H.A., Field, C.B. (2002). Grassland Responses to Global Environmental Changes Suppressed by Elevated CO_2. *Science 298*, 1987-1990.

Sitch, S., Huntingford, C., Gedney, N., Levy, P.E., Lomas, M., Piao, S.L., Betts, R., Ciais, P., Cox, P., Friedlingstein, P., Jones, C.D., Prentice, I.C., Woodward, F.I. (2008). Evaluation of the terrestrial carbon cycle, future plant geography and climate-carbon cycle feedbacks using five Dynamic Global Vegetation Models (DGVMs). *Global Change Biology 14*, 2015-2039.

Six, J., Bossuyt, H., Degryze, S., Denef, K. (2004). A history of research on the link between (micro)aggregates, soil biota, and soil organic matter dynamics. *Soil & Tillage Research 79*, 7-31.

Six, J., Carpentier, A., van Kessel, C., Merckx, R., Harris, D., Horwath, W.R., Luscher, A. (2001). Impact of elevated CO_2 on soil organic matter dynamics as related to changes in aggregate turnover and residue quality. *Plant and Soil 234*, 27-36.

Six, J., Conant, R.T., Paul, E.A., Paustian, K., 2002a. Stabilization mechanisms of soil organic matter: Implications for C-saturation of soils. *Plant and Soil 241*, 155-176.

Six, J., Feller, C., Denef, K., Ogle, S.M., Sa, J.C.D., Albrecht, A. (2002b). Soil organic matter, biota and aggregation in temperate and tropical soils - Effects of no-tillage. *Agronomie 22*, 755-775.

Skinner, R.H., Sanderson, M.A., Tracy, B.F., Dell, C.J. (2006). Above- and belowground productivity and soil carbon dynamics of pasture mixtures. *Agronomy Journal 98*, 320-326.

Sollins, P., Homann, P., Caldwell, B.A. (1996). Stabilization and destabilization of soil organic matter: mechanisms and controls. *Geoderma 74,* 65.

Soussana, J.F., Lüscher, A. (2007). Temperate grasslands and global atmospheric change: a review. *Grass and Forage Science 62*, 127-134.

Strömgren, M., (2001). *Soil-surface CO_2 flux and growth in a boreal Norway spruce stand. Effects of soil warming and nutrition*. Doctoral Thesis. Acta Universitatia Agriculturae Sueciae, Silvestria 220, Swedish University of Agricultural Sciences, Uppsala, ISBN 91-576-6304-1. 44 p.

Stevenson F.J. (1994) *Humus Chemistry: Genesis, Composition, Reactions*. John Wiley & Sons, New York.

Suseela, V., Conant, R.T., Wallenstein, M.D., Dukes, J.S. (2012). Effects of soil moisture on the temperature sensitivity of heterotrophic respiration vary seasonally in an old-field climate change experiment. *Global Change Biology 18,* 336-348.

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

Teixeira, E.I., Brown, H.E. (2012). *Climate change impacts on arable crop productivity*. In: The New Zealand Institute for Plant & Food Research Limited. Pp. 38.

Tisdall, J.M., Oades, J.M. (1982). Organic matter and water-stable aggregates in soils. *Journal of Soil Science 33,* 141-163.

Trumbore, S.E., Chadwick, O.A. & Amundson, R. (1996). Rapid exchange between soil carbon and atmospheric carbon dioxide driven by temperature change. *Science 272,* 393-396.

Tubiello, F.N., Soussana, J.F., Howden, S.M. (2007). Crop and pasture response to climate change. *Proceedings of the National Academy of Sciences 104,* 19686-19690.

von Lützow, M., Kögel-Knabner, I., Ekschmitt, K., Matzner, E., Guggenberger, G., Marschner, B., Flessa, H. (2006). Stabilization of organic matter in temperate soils: mechanisms and their relevance under different soil conditions- a review. *European Journal of Soil Science 57*, 426-445.

Wand, S.J.E., Midgley, G.F., Jones, M.H., Curtis, P.S. (1999). Responses of wild C4 and C3 grass (Poaceae) species to elevated atmospheric CO_2 concentration: a meta-analytic test of current theories and perceptions. *Global Change Biology 5,* 723-741.

Watt, M.S., Palmer, D.J., Kimberley, M.O., Höck, B.K., Payn, T.W., Lowe, D.J. (2010). Development of models to predict Pinus radiata productivity throughout New Zealand. *Canadian Journal of Forest Research 40,* 488-499.

Way, D.A., Oren, R. (2010). Differential responses to changes in growth temperature between trees from different functional groups and biomes: a review and synthesis of data. *Tree Physiology 30,* 669-688.

White, T.A., Campbell, B.D., Kemp, P.D., Hunt, C.L. (2001). Impacts of extreme climatic events on competition during grassland invasions. *Global Change Biology 7,* 1-13.

Xiao, C., Janssens, I.A., Liu, P., Zhou, Z., Sun, O.J. (2007). Irrigation and enhanced soil carbon input effects on below-ground carbon cycling in semiarid temperate grassland. *New Phytologist 174*, 835–846.

Yang, X.-M., Wander, M.M. (1999). Tillage effects on soil organic carbon distribution and storage in a silt loam soil in Illinois. *Soil and Tillage Research 52,* 1-9.

Zhou, X.H., Weng, E.S., Luo, Y.Q. (2008). Modeling patterns of nonlinearity in ecosystem responses to temperature, CO_2 and precipitation changes. *Ecological Applications 18,* 453-466.

5 Impacts of climate change on soil nitrogen cycling

Tony van der Weerden[1], Mark Shepherd[1], Murray Davis[2], Steve Thomas[3], Bryan Stevenson[4], Denis Curtin[3], Cecile de Klein[1]

[1]AgResearch, [2]Scion, [3]Plant and Food Research, [4]Landcare Research

5.1 KEY MESSAGES

Climate change impacts nitrogen (N) cycling through elevated temperature, eCO_2 and changing rainfall distribution and amount. These impacts are further modified by an increase in extremes: more hot days, greater frequency of drought and more storm events.

These conditions will interact to provide a net effect on N transformations and processes. In deliberating the likely effects, we also need to factor in the difference between response in an experiment and a larger scale response of the system (e.g. plant community structure). Experiments tend to provide information about the details of N processes but cannot really deal with interactions and complexity at the farm system level. Parsons et al. (2013) provide a salutary warning:

> *The complexity and time scales of response of this system defies understanding by observation and experiment alone, to the extent that attempts to manipulate the system without prior careful analysis of the potential outcomes, could prove at best ineffective, and at worst counter-productive.*

An assessment of the likely impact of climate change on N cycling and N losses in agricultural and forestry systems needs to consider:

- Direct effects on process rates in soils and plants,
- Indirect effects, e.g. on sources of N and C,
- Interactions between the above,
- Impacts of changed weather patterns on modifying the general trends in increased temperature and $eCO_{2.}$

This latter point is especially important; whilst we can extrapolate from experiments about general effects of temperature and eCO_2, the complexity derives from some of the extremes, most notably drought effects, which could negate any benefits from eCO_2 on productivity (Walthall et al., 2012). Furthermore, for pastoral systems we have not considered the contribution from risk of reduced animal performance due to hotter days under climate change.

This review showed that there are limited New Zealand specific datasets directly applicable to our farm and forest systems for assessing the potential changes of eCO_2 on N processes. All of the New Zealand specific data for pastoral systems is from the NZFACE experiment near Palmerston North in the Manawatu region (single site, one soil type, sheep grazed pasture). There is a risk of being too dependent on a single dataset. There are no specific local data for cropping or forest systems.

Table 5.1 summarises the estimated impacts of climate change on the N cycling processes for each ecosystem. The following assumptions are made:

- Dairy vs dry stock farming: we have used this as a guide to ‘intensity’, assuming dairy systems will be reliant on external N inputs (fertiliser and feed), which will not be sustained by potential benefits to legume production alone. In contrast, we have assumed that the dry stock systems will be driven primarily by N fixation, supplemented with N fertiliser on parts of the farms where economics allow.
- Atmospheric N inputs will increase as a result of slightly more rainfall. Given the low rates of atmospheric deposition in NZ, change in loads will be small. Increases in ammonia volatilisation may also increase atmospheric deposition of N.
- N fixation will increase because the balance of evidence suggests that eCO_2 favours legume growth.
- Increased N fertiliser inputs are assumed to be used to mitigate against PNL and are not limited by regulation; we have assumed legumes will be sufficient to mitigate PNL on dry stock farms (i.e. where N fertiliser inputs have traditionally been low).
- Excreta and effluent returns in pasture systems will increase as a result of more pasture growth.
- Increases in temperature and substrate availability will generally increase microbial activity so that mineralisation-immobilisation turnover (MIT) and nitrification may increase. The net effects on MIT are uncertain.
- An increase in the rate of ammonia volatilisation is driven by increased temperature and periods of drought. Sources (manure, urine, effluent, fertiliser etc.) may also increase.
- An increase in the denitrification rate is driven by increased temperature, mild winters and high water contents where plants become more water-use efficient and use less water, and due to periods of increased rainfall. Sources (manure, urine, effluent, fertiliser etc.) may also increase.
- Nitrate leaching will be a balance of other processes affecting the amount of mineral N remaining in the soil at times of drainage. There are insufficient data on leaching of dissolved organic nitrogen (DON).
- Removal of N in products (food and fibre) should increase where N and water are non-limiting due to eCO_2 fertilisation.

Table 5.1: Summary of the impacts of climate change on N cycling.

Soil nitrogen factor		**Production Sector**				**Justification**
		Cropping	Pasture intensive (e.g. dairy)	Pasture extensive (e.g. dry stock)	Forestry	
N inputs/ transfers	Atmospheric inputs	➚	➚	➚	➚	• Increase slightly due to slightly more rainfall and small increases in ammonia volatilisation.
	N fixation	➚	↗	↗	➚	• Increase due to balance of evidence suggesting eCO_2 favours legume growth.
	Fertiliser N	↗	↗	→	➚	• Increased inputs will be used to mitigate against PNL, limited by regulation. Assumed legumes will be sufficient to mitigate PNL on dry stock farms.
	Crop residues	➚	n/a	n/a	➚	• Increased forest productivity will increase residues at harvest
	Litter/ root exudation	➚	➚	➚	➚	• Increased due to increased NPP and legume growth.
	Effluent N	n/a	➚	n/a	n/a	• Increase in dairy systems as a result of more pasture production.
	Excreta N	n/a	➚		n/a	• Increase in dairy systems as a result of more pasture production.
Process rates	Net mineralisation				➚	• Increases in temperature and substrate availability will generally increase microbial activity, however net effects are uncertain. Evidence indicates an increase in forests.
	Nitrification	➚	➚	➚		• Likely to be a small increase due to small temperature increases, uncertain effects in forestry
Losses/ removal	Product	↗	↗	↗	➚	• Where N and water are non-limiting yield should increase due to eCO_2 fertilisation. More certain in forests.
	NH_3	➚ (minor)	➚	➚	➚ (minor)	• Increase driven by increased temperature and periods of drought. Sources (manure, urine, effluent, fertiliser etc.) may also increase. Effect on cropping likely to be small.
	N_2 & N_2O	➚	➚	➚	➔	• Increase driven by increased temperature, mild winters and periods of increased rainfall. Sources (manure, urine, effluent, fertiliser etc.) may also increase.
	NO_3	↗ ↘		↘	➚ ➘	• Balance of other processes affecting the amount of mineral N remaining in the soil at times of drainage. Will increase with increasing occurrence of drought and drought breaking rain. Will increase in forests where N-fixing shrubweeds present.
	DON					• Insufficient data on leaching of DON. Small increases possible where rainfall increases.

Key Direction of change and certainty of prediction:

➚ Overall, most likely to increase.
➘ Overall, most likely to decrease.
➔ Overall, most likely to remain unchanged.
➚➘ Could increase, remain unchanged or decrease.
\- Impacts are reasonably uncertain; therefore the direction of change cannot be predicted.

■ Reasonably certain of effects
▨ Neither certain nor uncertain
n/a Not applicable

5.2 INTRODUCTION

The N cycle is complex, with many interactions and feedback loops as N moves through the soil-plant or soil-plant-animal continuum (Figure 5.1). Processes are common between ecosystems but the relative size of fluxes will differ. Thus, although there are some commonalities in the likely response of N cycling to climate change, each ecosystem needs to be considered separately. Section 5.3 provides more commentary on the generic effects of likely climate change scenarios on these soil processes.

In this review we explore the impacts of climate change by focusing on the following four questions:

1. What are the impacts of climate change on N inputs and soil biological N processes (including immobilisation, mineralisation, nitrification and denitrification)?
2. What are the impacts of climate change on N losses (ammonia volatilisation, denitrification/nitrous oxide emissions and N leaching) from the pastoral, cropping and forestry sectors?
3. What are the potential consequences for production within the pastoral, cropping and forestry industries?
4. What are the impacts of climate change on future management of farm and forestry systems, and likely adaptations, including fertiliser management?

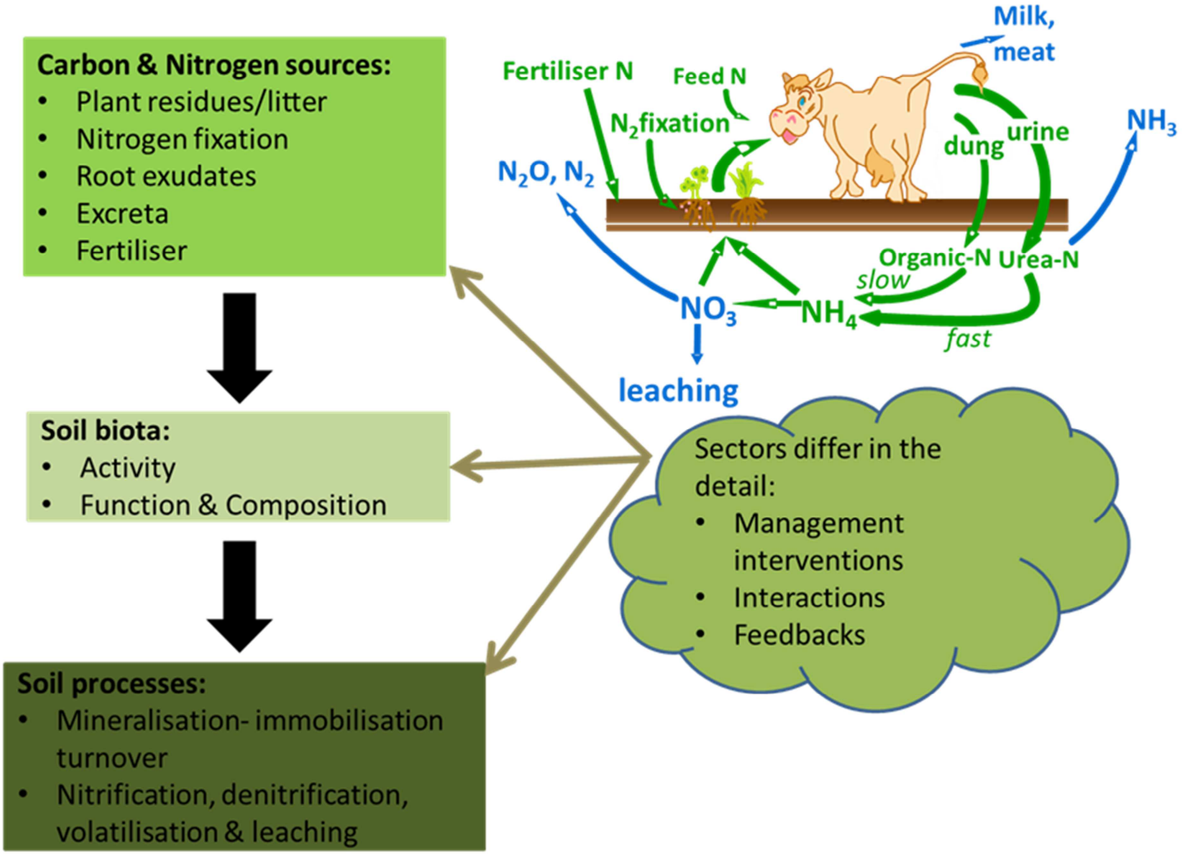

Figure 5.1: The N cycle, summarising the main sources and processes considered in this report.

We reviewed results from experiments exploring the effect of climate change on the following processes and transformations:

- N Inputs/transfers:
    - External: atmospheric inputs; N fixation; fertiliser N (though this will be a farm management *response* to climate change),
    - Recycled: crop residues; litter/root exudation; effluent N; excreta N.
- Process rates: mineralisation immobilisation turnover (MIT); nitrification.
- Losses/removal:
    - Gaseous: NH_3 volatilisation; denitrification (N_2 and N_2O),
    - Leaching of mineral N and dissolved organic N (DON),
    - N removal in food and fibre.

5.3 WHAT ARE THE IMPACTS OF CLIMATE CHANGE ON NITROGEN INPUTS AND SOIL BIOLOGICAL NITROGEN PROCESSES?

Effect on inputs

Plant material serves as the C and N source that drives the N cycling system (the ‘fuel’) through litter deposition, root exudation and root senescence (Hu et al., 2006) as well as via ingestion and excretion by grazing animals (dung, urine and, indirectly, effluent). Changes in rainfall may lead to small changes in atmospheric N inputs in rainfall (Parfitt et al. 2006).

Nitrogen fixation can also be a significant input in some systems. Conversely, increased pest and disease pressures can reduce potential plant productivity and yield, thereby influencing soil N cycling. Understanding the impact of climate change on plant composition and productivity is therefore an essential first step in predicting responses of N cycling processes to climate change.

Generic plant responses to climate change: implication for nitrogen

Elevated CO_2 increases photosynthesis in C_3 plants and decreases stomatal conductance and transpiration in C_3 and C_4 plants (Long et al., 2004) by, on average, 22% in crops and grassland species (Lenka & Lal, 2012). Some authors argue that increased soil moisture arising from lower transpiration increases plant growth in dry conditions and also affects soil processes. However, this assumes that effects observed at the individual leaf level also occur at the canopy, field and ecosystem level (Lenka & Lal, 2012); increased leaf area and/or increased soil evaporation due to increased temperatures might negate these effects (Reich et al., 2006).

Responses will not be fully expressed if other nutrients (or water) are limiting. For example, eCO_2 has been reported to increase N fixation in legumes. However, in two long-term experiments where this effect was short lived, the subsequent decline in N fixation was associated with shortages of other nutrients: molybdenum (Hungate et al., 2003); or low soil P availability (Niklaus & Körner 2004).

Species-specific responses to eCO_2: implications for nitrogen

Tubiello et al. (2007a) provided a synthesis of crop response data to eCO_2. They reported an increase in photosynthesis under optimal conditions of 30-50% in C_3 plants and 10-25% in C_4 species from a doubling of atmospheric CO_2 concentration, a crop yield response of 10-20% for C_3 species and 0-10% for C_4 species (CO_2 increased from 380 to 550 ppm), and pasture yield response of ca. 10% for grasses and 20% for legumes.

Tubiello et al. (2007a) also postulated that future CO_2 levels would favour C_3 species but that warming would favour C_4 species, such that there is uncertainty about net effects. Dodd et al. (2010) suggest that other factors also need to be considered: competitiveness of C_4 species with other species, selection effects of grazing animals and soil biogeochemical feedbacks. They concluded that where C_3 species dominated, competition from C_4 species would be limited by progressive N limitation (PNL, discussed in detail later); where C_4 species dominated, their ability to benefit from warming would be limited by PNL unless fertiliser was added.

Legumes, including white clover, can benefit from eCO_2 by increasing the number and mass of root nodules (Zanetti et al., 1996). The rhizobial population composition can also be influenced by eCO_2. Thus, eCO_2 can lead to an increase in N inputs into agricultural systems if legumes are present. An increase in legume biomass under eCO_2 was noted in the NZFACE experiment (Ross et al., 2004). In cropping systems, eCO_2 increases grain N accumulation in legumes. In a meta-analysis Lam et al., (2012) found that eCO_2 increased N fixation by 38%, accompanied with increases in nodule number (+33%), nodule mass (+38%) and nitrogenase activity (+37%). However, more N was taken off as grain than fixed by the crop (Lam et al., 2013; Lam et al., 2012), requiring more soil and fertiliser N inputs.

Elevated CO_2 has been shown to enhance productivity or dominance of the understory of N-fixing woody species *Robinia* and *Eleagnus* in forests (Mohan et al., 2007, Norby & Zac, 2011) which would be expected to increase N-fixation. Thus, N inputs from important understory N-fixing species such as gorse (*Ulex europeus*) and broom (*Cytisus scoparius*) and other N-fixing species that occur in the understorey of New Zealand forests (eg *Lotus pedunculatus*, *Coriaria arborea*) should increase due to eCO_2, provided their growth is not limited by soil moisture or other nutrients.

Effects of residues on nitrogen processes

To understanding how plants respond to eCO_2 and temperature change it is important to understand how soil inputs will be affected by residues. The quantity of surface residue is likely to increase in eCO_2 environments where more biomass is produced (Prior et al., 2004). Dieleman et al. (2012) concluded that foliar N concentrations in living plants decrease significantly in combined eCO_2 and warming treatments, which may lead to an increase in C:N ratios of resulting residues. Taub & Wang (2008) reviewed 10 hypotheses for the mechanism(s) but tended to favour a combination of dilution of N in plant material due to increased C compounds derived from photosynthate and decreases in N uptake.

In most arable systems, the crop harvest index (i.e. the harvested fraction of total plant biomass) is marginally affected by eCO_2 (Pleijel& Uddling, 2011). This will mean that as grain yield increases, the amount of residue, either on the surface or incorporated into soil, will also increase by the same proportion. Based on an international meta-analysis, the effects on C:N ratios may be crop specific. From their analysis Lam et al. (2012) found that, overall, C:N ratios in C3 crops increased by about 16%, while those in legumes increased by about 8%. There was no effect of eCO_2 on C:N ratios in C4 crops. However, changes in C:N ratio may be nil or small where there is a strategy to attain increased yields under eCO_2 or to ensure high N grain concentrations to meet specific quality objectives.

Initial thoughts were that the higher C:N ratio of plant material and an increased content of lignins and phenolic compounds under eCO_2 would reduce the quality of litter and its rate of decomposition. Reviews of the data, however, have not shown this to be the case (Hu et al., 2006; Reich et al., 2006). Hu et al. (2006) suggest that eCO_2-induced changes in quantity of C would dominate decomposition and N mineralisation and that any changes in substrate N content may play a secondary role.

Effects of climate change on soil biological processes

Much of the N cycle is mediated by soil floral/faunal communities (the 'engine') and we need to consider the direct (warming) and indirect (warming, eCO_2, Progressive N limitation, extreme weather events) effects of climate change on the activity and composition of these populations (Pritchard, 2011).

Warming

Temperature effects on N cycling processes are relatively well understood: an increase in temperature on its own will tend to increase transformation rates unless something else is limiting (e.g. moisture). Thus, soil microbial activity, mineralisation-immobilisation turnover (MIT), nitrification, volatilisation and denitrification rates would all be expected to increase within temperature increases predicted in the current climate change scenarios.

eCO_2

In contrast to temperature, eCO_2 is not reported to have direct effects on process rates: the effects of eCO_2 will be indirect, e.g. on residue amount and quality, thus affecting C supply and the biomass. These interactions provide the net effect of climate change, further modified by other environmental or management changes that may be direct or indirect effects of climate change.

Soil microbes utilise C in the soil as an energy source, so if eCO_2 increases soil C inputs via litter deposition or root exudation, soil biomass is likely to respond positively. Hu et al. (2006) reviewed up to 40 papers and found an increase in biomass C reported in 19 of 40 observations (no negative effects, average increase 26%); and an increase in biomass N in 12 of 27 observations (no negative effects, average increase 24%). A similar trend was also found with microbial respiration. Recent observations from a FACE experiment with loblolly pine (*Pinus taeda*) in low N soil (North Carolina, USA) indicate that long term CO_2 enrichment may stimulate microbial activity and soil organic matter decomposition and increase soil N mineralisation (Phillips et al., 2011; Drake et al., 2011). In contrast, a long-term open top chamber study with *P. radiata* and red beech (*Nothofagus fusca*) in New Zealand with eCO_2 increased microbial activity but did not affect net N mineralisation or nitrification (Ross et al., 2006). However, the most consistent effect was increased soil heterotrophic respiration in the vast majority of experiments in response to eCO_2 and warming (Dieleman et al., 2012), although some degree of acclimation is likely (Pritchard, 2011). Zak et al. (2000) estimated from the published data that soil respiration increased by an average of 51% under grassland (elevated CO_2 concentrations were typically ambient CO_2 concentration + 360 ppm). Pritchard (2011) explained this by greater root exudation under eCO_2, which 'primes' the food web leading to higher rates of organic matter decomposition.

FACE experiments have demonstrated that eCO_2 has only a small effect on bacterial and fungal community composition in forests, despite increased below-ground litter inputs (Norby and Zac 2011). Further, although the activity of the microbial community and N cycling rates may be increased by eCO_2, the magnitude of response is much less than that observed due to spatial variation in soil properties, temporal changes in environmental conditions and plant community composition.

There has also been some suggestion that global warming could favour soil fungi over bacteria (Pritchard, 2011), mediated by a widening of C:N ratio of litter and root exudates (Hu et al., 2006), thereby altering the microbial community structure (Hu & Zhang 2004). Reviews by Hu et al. (2006) and Pritchard (2011) concluded that mycorrizal growth will increase.

Combined warming and eCO_2 experiments are rare. The majority of experiments have focused on eCO_2. However, even though the combined effects of eCO_2 and increased warming, for example, are rarely additive, combined temperature and eCO_2 responses have been shown to be similar to eCO_2-only treatments (Dieleman et al., 2012). Uncertainties in response increase with combined effects.

Progressive N limitation (PNL)

The impact of PNL (Hu et al., 2006) on a system’s ability to express a CO_2 fertilisation effect needs to be understood. N availability declines under eCO_2 due to increased N uptake, while warming typically increases soil N availability due to increased mineralisation (Dieleman et al., 2012). A major uncertainty in predicting effects of climate change on soil N cycling is estimating the relative effects of climate change on net primary production, i.e., will a CO_2 fertilisation effect be sustained?

Systems that rely on low external N inputs (and, by inference, where legume contribution through fixed N is also low) are at risk of a yield decline due to PNL (Hu et al., 2006; Reich et al., 2006). The nature and magnitude of external N inputs, the initial N status of the ecosystem and changes in soil C stocks will critically determine if and when PNL occurs; PNL is unlikely to occur in ecosystems that receive substantial external inputs (Hu et al., 2006).

While in the short term eCO_2 stimulates growth and increases the N (and C) stock in plant biomass and in soil organic matter, over longer periods productivity may become limited by an insufficient N supply due to immobilisation by plant biomass and, where soil C stocks are increasing, by soil organic C, resulting in a negative feedback to plant growth.

Thus, changes in soil C stocks influence the likelihood of PNL occurring. It is uncertain whether soil C stocks will increase, remain constant, or decline over time. Lu et al. (2013) assessed the effect of warming on ecosystem C cycling through a meta-analysis of 130 studies, and concluded that increases in plant derived C influxes offset any increase in C losses to the extent that ecosystems may be a small C sink. However, this analysis did not include the combined effects of increasing temperature and eCO_2.

Modelling of the effect of eCO_2 and increasing temperatures on soil C stocks in a forestry system suggests that soil C stocks are more likely to remain constant or decline with time (Kirschbaum et al., 2012). In the FACE experiments, PNL has so far only developed at one of five forest sites, despite the forests being N limited (Norby & Zac, 2011). Under grassland systems with low external N inputs (e.g. low N fixation by legumes in grass/clover swards) the effect of PNL can be variable (Ross et al., 2004; Hu et al., 2006) and temporary, as suggested by the “re-setting” the grassland system with drought-breaking rain (Newton et al., 2010). It is critically important to remember that given many of the eCO_2 experiments focus on low N systems, it is possible that these trends may differ where N supply is less limited (e.g. by the addition of fertiliser N at rates > 200 kg N/ha/year).

PNL may induce a management response that could impact on the overall effect of climate change on N cycling processes. As an example, consider ammonia volatilisation, which is mainly a chemico-physical process. Increased temperature will increase volatilisation, as will lengthier periods of drought, if there is insufficient moisture to facilitate transfer of N sources into the soil matrix and/or transform ammonia into other N forms. However, in order to assess the net effects of climate change on this response, sources also need to be considered, e.g. fertiliser inputs, dung, urine and effluent deposition. If the management response is to apply less N fertiliser in times of drought, the net effect on volatilisation from this source could be a decrease; if more is applied through the year to counteract PNL, the volatilisation would increase. If a response is to switch from urea to a nitrate based fertiliser then volatilisation would similarly decrease. If more pasture was grown with more grazing animals supported, then volatilisation from more excreta would increase. Furthermore, NH_3 volatilised from neighbouring systems upwind may be deposited downwind, which may balance the effect of increased volatilisation rates. Thus, extrapolation to a system response needs to factor in all considerations.

Extreme weather events

Changing temperature and rainfall patterns and intensities can have either a negative or a positive impact on N mineralisation. Longer periods of high soil water deficits following increases in the frequency of hot, dry days will impact N transformations in the absence of irrigation. Effects include summer droughts, which will reduce growth and produce a flush of mineralisation on rewetting (Xiang et al., 2008; Newton et al., 2010).

Major forest disturbances such as dieback, fire, and wind-throw that create forest gaps generally lead to short-term increases in N mineralisation (Atiwill &Adams, 1993) and potential for N-fixing shrubweed invasion, so increased climate change-induced disturbance should increase soil N mineralisation. Fire may also increase nitrification (Knoepp & Swank, 1995), however moderate increases in fire frequency may not affect long term ecosystem N storage (Smithwick et al., 2009). Drought effects on soil processes are uncertain. Although microbial activity will be inhibited in dry soils, death of microbes may result in increased N availability (Rennenberg et al., 2009).

Whilst clearly the long-term annual effects of changes in climate are important, we do also need to consider the impacts of these extreme effects. Walthall et al. (2012) cite the uncertainty of precipitation and the extreme (hot) temperatures as factors that will limit agricultural and forestry productivity and therefore override the effects of average changes in climate on N cycling processes in extreme seasons.

5.4 WHAT ARE THE IMPACTS OF CLIMATE CHANGE ON NITROGEN LOSSES FROM THE PASTORAL, CROPPING AND FORESTRY SECTORS?

Elevated CO_2 and temporal and spatial rainfall variability will be key factors affecting NH_3 volatilisation, denitrification, N_2O emissions and N leaching responses.

Ammonia volatilisation

Ammonia volatilisation in pasture systems predominantly occurs from applied fertiliser, deposited urine and effluent/manure. Increased temperature increases ammonia volatilisation; increased precipitation decreases volatilisation; increased drought increases volatilisation (Meisinger, 2000). If there is an increase in %N in the diet due to increased clover content, a greater proportion of a grazing animal’s dietary N may be partitioned to urine, resulting in a greater volatilisation risk (Allard et al., 2003). Ammonia volatilization from urine, effluent and inorganic fertiliser applications may increase due to to increased temperature and drought (Butterbach-Bahl & Dannenmann, 2011), which may influence associated emission factors. Similarly, increased N fertiliser inputs, particularly urea fertiliser, to maximise eCO_2 effects on crop production will increase NH_3 volatilisation. Increased volatilisation losses from forests may also occur if foresters increase urea usage to avoid development of PNL in response to CO_2 fertilisation of forests (Kirschbaum et al., 2012).

Denitrification/N_2O emissions

Denitrification is affected mainly by soil C availability, soil oxygen levels, N supply and temperature. Below-ground C inputs will increase under eCO_2, leading to changes in root biomass, depth, distribution and exudation (Cheng & Johnson, 1998; de Graaff et al., 2007; Pritchard & Rogers, 2000) and soil respiration (Thomas et al., 2000). Changes in soil organic matter cycling are dependent on soil N availability. Decomposition of soil organic matter increases at high N, while at low N concentrations it decreases (Cheng & Johnson, 1998). It has been proposed that increased C inputs into the soil could stimulate soil respiration (and increase soil oxygen consumption), increasing periods or sites of soil anoxia and thereby stimulating denitrification (Butterbach-Bahl & Dannenmann, 2011). Increased precipitation and warmer temperatures may increase denitrification and N_2O emissions because of increased microbial activity due to increased soil moisture contents occurring for longer periods (Butterbach-Bahl & Dannenmann 2011; Brown et al. 2012), with some evidence suggesting a decrease in the N_2O:N_2 ratio (Smith 1997). Furthermore, deeper rooting and root exudation could lead to increased subsoil denitrification that could reduce the risk of nitrate leaching (Butterbach-Bahl and Dannenmann, 2011). In contrast, prolonged periods of drought may decrease N_2O emissions (Hartmann & Niklaus 2012).

Meta-analyses of 49 international published studies of agricultural and natural systems suggest that eCO_2 increased N_2O emissions by 19% mainly due to increased root biomass (van Groenigen et al., 2011). A larger increase of 27% was determined by Lam et al. (2012) from their meta-analysis of 127 studies of N dynamics in grain crop & legume pasture systems. Greatest responses in N_2O emissions occur in higher N input systems while responses are consistently low and non-significant in unfertilised trials (Dijkstra et al., 2012). Indirect eCO_2 effects have been attributed to: more labile N in the rhizosphere; more soil moisture due to effects of eCO_2 on increased water use efficiency (Arnone Iii & Bohlen, 1998) and stimulation of soil biological activity. There is a suggestion that eCO_2 may decrease the N_2O:N_2 ratio (Baggs et al., 2003).

Increasing temperature has inconsistent or relatively small effects on N_2O emissions (Dijkstra et al., 2012). These researchers suggest that the size of the relative responses to eCO_2 and temperature reflects the differences in treatments applied; they observed that in most studies eCO_2 concentrations were doubled whereas soil temperature increases were relatively small. There is little experimental information on the combined effects of eCO_2 and increasing temperature and where experiments (of limited length and replication) have been conducted no significant interactive effects have been observed (Dijkstra et al., 2012).

N_2O losses from forestry are low compared to other land uses, particularly grazed pastures (Saggar et al. 2008), with little evidence to suggest that climate change will impact on emission levels from forestry systems. Warming has not been found to have large direct effects on N_2O emissions (Barnard et al 2005). In a study by Butler et al. (2012) of deciduous forest in Massachusetts USA, soil warming (5^{o}C for seven years) did not have any discernible effect on N_2O fluxes. To determine the effects of eCO_2 on N_2O emissions, Barnard et al. (2005) conducted a meta-analysis of 20 experiments which included both forest and herbaceous systems. N_2O fluxes were not significantly altered by eCO_2 when measured either in the field or in the laboratory.

Nitrogen leaching

Climate change will impact on N leaching by affecting the source (i.e. the quantity of N available for leaching) and the transport of this N (movement through the soil/drains). The process of leaching will be therefore driven by drainage (precipitation) and size of the available N pool. Although Larsen et al. (2011) found that precipitation increased N leaching, estimates of the effect of climate change on leaching are uncertain due to the competing processes of source vs. transport (Stuart et al., 2011). For example, Torbert et al. (1996) observed lower leaching losses below soybean and sorghum grown in elevated CO_2 compared to ambient CO_2, and attributed this largely to increased uptake of soil N.

While increasing temperatures are expected to increase the rate of soil N processes such as mineralisation and nitrification, there may be no significant effect on N leaching (Turner & Henry, 2010) for pastoral, arable and forestry systems. However, eCO_2 can lead to higher soil water contents through improved plant water use efficiency, which may lead to increased risk of leaching losses or provide conditions more conducive to denitrification (Prior et al., 2011). This may be compensated by increased plant production where water availability may otherwise be limiting. The soil mineral N pool may be decreased by eCO_2 (Zhang et al., 2005) due to increased uptake of soil N (Torbert et al., 1996) . Subsoil denitrification promoted by increased labile C inputs in larger amounts and deeper in the profile may reduce N losses (Thomas et al., 2012). In low fertility systems the benefits may be greater due to soil N immobilisation reducing the amount of available NO_3 for leaching.

It is probable that annual and seasonal leaching patterns will be highly responsive to temporally-spatially (regional) variable rainfall patterns. We might expect intra- and inter-annual spatial and temporally variable patterns that are similar to the crop responses predicted by modelling approaches using NIWA climate scenarios in Figure 5.3 (Teixeira & Brown, 2012). Modelling approaches may offer a means to better understand the risk of leaching under different climate change scenarios (Borgesen & Olesen, 2011; Sjoeng et al., 2009), but these have not been applied for future New Zealand climate scenarios. For example, increased temperatures may lead to longer growing seasons when plants are actively extracting water and taking up soil N.

As noted above, a greater proportion of grazing animals’ dietary N may be partitioned to urine if there is an increase in %N in the diet due to increased clover content (Allard et al., 2003), which may enhance N leaching. However, modelling of a dairy farm under a range of climate change scenarios (eCO_2, elevated temperature, drought or increased rain) suggested a decrease in N leaching, but increases in denitrification and volatilisation (Dueri et al., 2007).

In regions that are already wet, additional precipitation may not have a large effect on the transport of available N through the soil profile, since a large proportion of it may have been leached anyway. Effects will be larger where, without climate change, there was insufficient drainage to result in N leaching through the soil profile.

As well as nitrate forms, N can leach as dissolved organic N (DON). Wachendorf et al. (2005) measured c. 50 kg N/ha as DON leached from cattle urine patches. According to van Kessel et al. (2009), DON is the missing N in nutrient budgets in that it is a much overlooked source of N loss. They reported that 26% of soluble N can be as DON, on average. DON is usually the major form of N leached from native forests and exotic forests planted on ex-pasture sites (McGroddy et al., 2008; Davis et al., 2012). Losses from New Zealand soils have been significant, with up to 117 kg N/ha being measured from shallow lysimeters (Ghani et al., 2007; 2010). Further work is required to understand the impacts of moisture, temperature and vegetation type (e.g. C4 grasses) to help determine the impact of climate change on losses of DON.

Compared to other land uses, N leaching losses from forestry are generally low (Binkley et al., 1999; Elliot et al., 2005; Larned et al, 2004). If forest growth increases in response to increased CO_2 concentrations, demand for N by both the crop and soil microbes should also increase. This increase in demand should in turn counterbalance any increased N that may become available because of soil warming-enhanced N mineralisation, and N leaching should not increase above present low levels.

In some forests, N-fixing species may become an important part of the vegetation after forests are harvested and before full site occupancy by the next rotation crop, for example on sand dunes where *Lupinus arboreus* may be present, on other soils where gorse and broom occur, or at some ex-pasture forest sites where legumes such as *Trifolium* or *Lotus* were present in the pasture. N leaching from forests with a pasture or N- fixing shrubweed history is higher than from forests without such a history (Parfitt et al 2002; Quinn & Ritter, 2003; Davis et al., 2012). In these situations increased atmospheric CO_2 concentrations may be expected to increase productivity and N-fixation by N-fixing species where they develop after harvest (Mohan et al., 2007; Norby & Zac, 2011). Thus, where N-fixing species are an important part of the inter-rotation vegetation, NO_3^- leaching may be expected to increase. Any increase in leaching is most likely to occur when N demand by the forest crop is low as in the latter part of the rotation, and immediately after harvest (Parfitt et al., 2002; Quinn & Ritter, 2003). For most forests, where N-fixing species do not form an important part of the inter-rotation vegetation, NO_3^- leaching is unlikely to be affected by CO_2 enrichment.

Climate change may lead to an increase in major forest disturbances such as dieback, fire, and wind-throw that create forest gaps resulting in short-term localised increases in N mineralisation (Atiwill & Adams, 1993). Enhanced NO_3^- leaching is likely to occur in such gaps up until the time the gaps are revegetated by weeds or by re-planting (Parfitt et al., 2002). Gap revegetation is usually rapid so N losses from such gaps are likely to be short term in nature.

5.5 WHAT ARE THE POTENTIAL CONSEQUENCES FOR PRODUCTION WITHIN THE PASTORAL, CROPPING AND FORESTRY INDUSTRIES?

Pastoral systems

The key features of the pastoral system that differentiates it from the other ecosystems are:

- Returns of N (and C) are spatially variable across a paddock (dung, urine), so we need to consider the likely impacts of differential N (and C) supply on processes (Figure 5.2).
- Effluent/manure and fertiliser N can be applied on top of the dung and urine patches.
- Animals mediate the N returns in dung and urine and we need to consider the impacts of changes to the composition of ingested forage on these N returns (Allard et al., 2003).

- There is a heavy reliance on legumes to fix and supply N in some systems.
- There are temporal fluctuations between legume and grass populations, with this co-existence of the two species adjusting to fluctuations in soil mineral N supply (Schwinning & Parsons, 1996).
- Sward composition can change with time, because of pasture management and the environmental conditions.
- Different enterprises (usually associated with different landscapes) vary in the size of N process fluxes, although the transformation processes are generally the same.
- Animals and animal products are the measure of productivity from pastoral farms: any effects on animals need to be considered when considering performance of the pastoral sector under climate change.

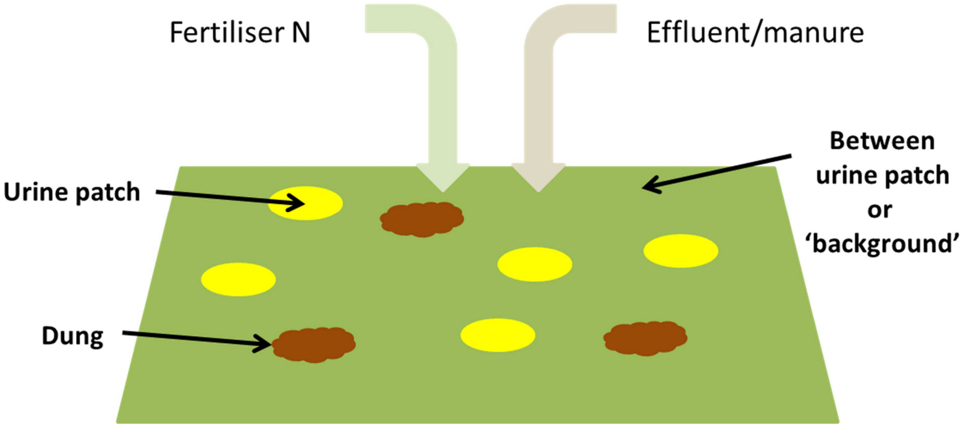

Figure 5.2: Representation of the spatial variation of nitrogen sources in a paddock.

Progressive N limitation (PNL) is often cited as a potential issue under $eCO_{2,}$ and this may well be the case in systems relying on low external N inputs, or in the absence of legumes. For example, in monocultures of *L. perenne*, response to eCO_2 was only apparent at high N input (Schneider et al., 2004). This response was caused by removing the N limitation.

However, in legume/grass pastures, PNL may be less of an issue. The interaction between grass and clover populations is complex but follows an N-based competitive trade-off between the two species (Schwinning & Parsons, 1996); when soil mineral N is low, clover is favoured and when high, grass is favoured. Through N cycling, the two species establish an intermediate level of soil mineral N where they can both co-exist. Thus, understanding the long-term effects of climate change on this interplay between grass and legumes is critical to understanding the consequences for production in New Zealand pastoral systems.

Experiments suggest that global warming will favour legume growth and N fixation (Zanetti et al., 1996; Ross et al., 2004). Thus, a change in the plant community structure, with an increase in legumes, could supply additional N to the system. Tubiello et al. (2007a) postulate that N limitation might be removed by an increase in biological N fixation under eCO_2. Legumes increased in the NZFACE experiment, although this appeared insufficient to completely remove a PNL effect (Newton et al., 2010).

If PNL can be overcome by sufficient N addition, productivity could increase, which, in turn, will lead to increased grazing and N removal in products such as milk, meat and wool. Pasture yield may increase by ca. 10% for grasses and 20% for legumes under eCO_2

(from 380 to 550 ppm; Tubiello et al., 2007a). Meta-analysis (Long et al., 2004) suggests on average a 20% yield response from eCO_2 for C_3 species and a response of <5% for C_4 species. There is uncertainty about net effects because eCO_2 levels would favour C_3 species, but warming would favour C_4 species (Tubiello et al., 2007a). Other management factors will influence competition between species (Dodd et al., 2010), as well as greater cyclical variation over time in N fixation and N supply in legume/grass pastures.

An example of additional N supply modifying PNL was demonstrated by Newton et al. (2010) in the NZFACE experiment. In this case there was evidence of pasture yield decreasing over a 4 year period, but this N limitation was broken after a severe drought that provided a flush of mineral N. There was evidence that PNL again started to occur in the years after the drought/rewetting effect.

Dieleman et al. (2012) concluded that foliar N concentrations in plants decrease significantly in combined eCO_2 and warming treatments. However, when scaled up from individual plant species to the plant community, this dilution effect has not always been noted because the composition of the plant community also responds to eCO_2. Reich et al. (2006) also suggest that any effects of eCO_2 on decomposition rate are more likely to be due to eCO_2 altering plant community composition. Indeed, the NZFACE experiment noted a reduced N concentration in individual species in the sward, but also a change in species composition under eCO_2 with a move to species such as legumes with a higher N content (Allard et al., 2003). While this offset the lower N concentration in the forage tissue, it led to greater partitioning of N into urine by grazing livestock, which has implications for N leaching and volatilisation.

Even if above ground biomass increase is limited by PNL, eCO_2 stimulation of photosynthesis often results in increased allocation of C below ground (Hu & Zhang, 2004). This has been attributed variously to an increase in root growth, root exudates and C supply to symbionts. This will serve as a useful C source for soil biological activity.

Animal health and performance

An assessment of the impact of climate change on animal health and performance was outside the scope of the report. However, the animal is obviously integral to the pastoral system, and West (2003) suggests that the effect of heat stress on animal performance could be a signicant impact of climate change. With the climate change scenario for New Zealand suggesting a 50% increase in hot days, some discussion on climate change effects on animal health is warranted.

Walthall et al. (2012) suggests that under climate change scenarios in the US, animal productivity will be reduced. In turn, N cycling in grazed pastoral systems may be reduced. Certainly, studies suggest that warming in the tropics and in the subtropics during warm months has adverse effects on livestock reproduction and production, including reduced animal weight, decreased dairy production, and less feed conversion efficiency (Klinedinst et al., 1993). However, results are mixed for impacts in temperate and cooler regions and it is suggested that the health and performance of forage-fed livestock may improve (due to more forage). Also, warming during the cold periods in temperate areas may be beneficial to livestock production, which could see N cycling through the soil-plant-animal system increase in southern South Island. Campbell et al. (1996) suggest that productivity of dairy farms might be adversely affected by a southward shift of undesirable subtropical grass species, such as *Paspalum dilatatum*, which currently affect the upper North Island.

Effects of extreme weather on crop/forage productivity

Similarly, Walthall et al. (2012) cite the uncertainty of precipitation patterns and the likely increase in droughts as key factors that will reduce productivity under climate change in the US. Thus, whilst the scope of this report is on soil N processes, this risk to production and N inputs needs to be carefully considered.

Arable systems

Globally, climate impacts on crop production are geographically heterogeneous. Overall, larger crop yield losses are projected for the tropics than for temperate and mid-latitude countries (Fischer, 2009; Fischer et al., 2008; Parry et al., 2005). Uncertainties in current projections include the magnitude of eCO_2 fertilisation effect and yield responses to temperature, soil nutrient supply, soil water availability and pest damage at different scales - farm, regional and global – where different factors interact, and responses to extreme weather events rather than to average changes in climate (Long et al., 2005; Tubiello et al., 2007a; Tubiello et al., 2007b).

For New Zealand in the 2040s, without considering extreme events and biotic stresses, modelling studies indicate relatively mild yield losses and potential for increases in crop yields in response to higher temperatures and eCO_2 (Teixeira & Brown, 2012). Crop responses were modelled using the NIWA high carbon world scenario (A2) and 'rapidly decarbonising' world scenario (B2) for between 2030 and 2049 for the four major cropping regions (Canterbury, Southland, Hawke's Bay and Waikato). Average temperature increases ranged from 0.9 to 1.5°C for the period, and rainfall changes ranging from -14% to +34% in a given season and location. Impacts varied largely depending on location, management and crop type (Figure 5.3).

By 2040 wheat and barley yields were predicted to increase by about 12%, with largest increases in Canterbury and Southland (Figure 5.3). Potato, field peas and maize yield responses were more variable and showed more frequent potential for decline in some scenarios. Positive yield responses were due to acceleration in canopy expansion due to higher temperature, increased photosynthesis due to CO_2 fertilisation and higher temperatures, and longer cycle lengths for winter forages. Negative responses included early flowering, reducing the period for intercepting solar radiation and shortening growth cycles, and water stress due to reduced rainfall during some seasons. For example, potato yields in Waikato were predicted to decline by 5 to 20% in most scenarios due to increases in temperature-induced shortening of the crop cycle, with a consequent reduction in intercepted photosynthetic radiation. Comparison of current and future scenarios suggests that current photosynthetic conditions for growing potatoes in the Waikato are close to optimum, i.e. there were no improvements in radiation use efficiency with increased temperature. In general, greatest benefits occur in more southerly regions, particularly for temperate cereals, green feed and forage brassicas. For harvested crops such as grain and potatoes, this is an N removal from the soil and farm system. Predicted changes in rainfall patterns, and therefore soil water availability, had a large impact on crop yield and inter-annual variability. This was most pronounced with crops grown on shallow soils (Figure 5.3). Yield responses to irrigation also varied widely between regions with responses ranging from 5% in Southland (heavy soil) to 60% in Canterbury (light soil).

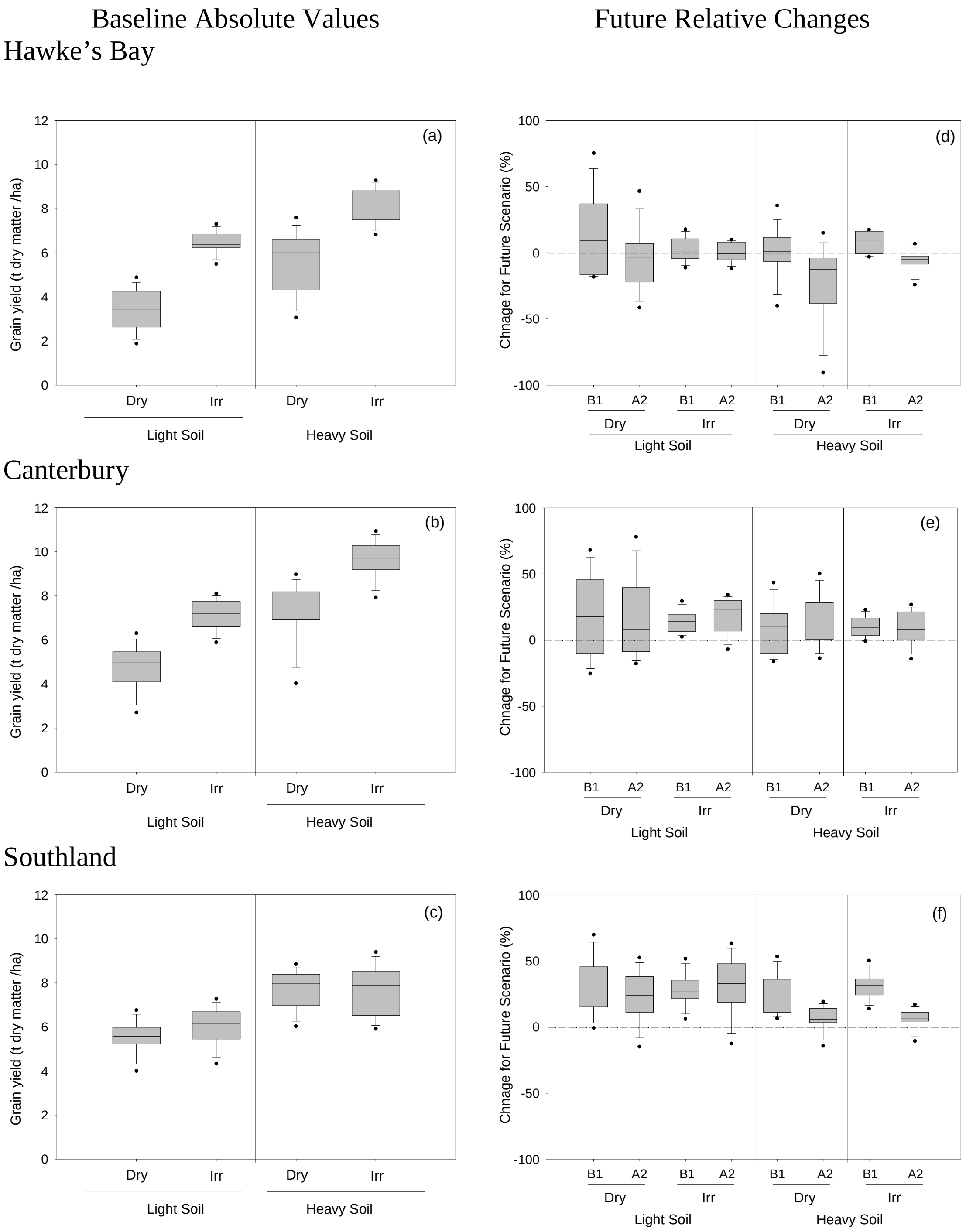

Figure 5.3: Wheat median grain dry matter yield for baseline climate from 1 January 1980 to 31 December 1999 and percent changes for 20 year simulations considering two future climate change scenarios (high carbon (A2) and 'rapidly decarbonising (B1) worlds), four locations, two soil types and two water supply conditions (dryland or irrigated) (Source: Teixeira & Brown, 2012).

Forestry systems

Increasing CO_2 concentrations are expected to increase forest productivity (Bazzaz & Miao, 1993; Ceulemans et al., 1999; Norby & Zac, 2011; Zac et al., 2011), through increased photosynthesis and reduced stomatal conductivity, and an increase in water use efficiency (Silva et al., 2010). Increasing CO_2 concentrations may also lead to increased root growth (Ceulemans et al., 1999; Norby & Zac, 2011) and deeper rooting systems.

As a consequence of increasing underground plant and microbial biomass stocks in CO_2-enriched environments, more N will become immobilised, which may lead to progressive N limitation (PNL) in unfertilised and infertile ecosystems (Luo et al., 2004; Dieleman et al., 2012; Garten et al., 2011; Melillo et al., 1993; Norby et al., 2010; Norby & Zac, 2011). However, increased N availability arising from enhanced N mineralisation caused by soil warming may counteract such limitation. A meta–analysis which included 14 studies (50% forest or woody species), where the combined effects of warming and eCO_2 were examined, showed that on average N immobilisation was counterbalanced by increased organic matter mineralisation, resulting in little overall change in N availability relative to control treatments (Dieleman et al 2012). The analysis also showed that foliar N concentrations declined under combined eCO_2 and soil warming treatments, indicating that at N limited sites PNL is ultimately likely to occur.

With declining N concentrations in plant tissues, C:N concentrations in forest residues (thinnings, prunings, harvest residues) will increase and such residues will take longer to decompose and immobilise increased N in the process, thus contributing to PNL development. In the FACE experiments, PNL has so far only developed at one of five forest sites despite the forests being N limited (Norby & Zac, 2011). Several possible reasons for this have been suggested, including that insufficient time had elapsed for an effect to develop, increased soil exploration by fine roots, stimulation of mineralisation by fungal activity, and the type of mycorrhizae. This last was suggested because the experiment where PNL became apparent was with *Liquidambur* which forms arbuscular mycorrhizas, in contrast to the remaining sites where the species form ectomycorrhizas.

A difficulty with a number of forestry studies is that large step changes in soil temperature of around $5^{o}C$ have been used, so their relevance to the New Zealand situation, where an approximately $1^{o}C$ temperature rise is expected, is questionable. However in a modelling study based on *Pinus radiata* growing under a Canberra (Australia) climate, Kirshbaum (1999) found that increasing temperature by $2^{o}C$ increased N mineralisation rates, and that the increase was qualitatively similar when conditions were changed gradually or as a single step change.

In New Zealand, climate change is predicted to increase wood production by an average of 19% by 2040 and 37% by 2090 (Kirschbaum et al., 2012). The response is expected to be due more to eCO_2 concentrations than to temperature or moisture effects. Any response will be dependent on maintenance of the present soil fertility levels in plantations, which tends to be generally adequate, although it may be locally limiting. Should increased productivity lead to development of PNL, wood density is likely to increase because N fertilisation and legume presence has been shown to reduce mean, early and late wood density (Beets et al., 2001).

Young *Pinus radiata* is sensitive to moisture competition, for example broom has been shown to reduce productivity of two-year-old pine by 10-fold at a dryland site (Watt et al., 2003). Enhanced growth of N-fixing shrubweeds with CO_2 enrichment (Mohan et al., 2007; Norby & Zac, 2011) may therefore reduce forest productivity because of increased competition for moisture. However reduced stomatal conductivity and improved water use efficiency of both crop and weed species should lead to higher soil moisture contents and partially counteract the increased competition. Because increased N may reduce wood density (Beets et al., 2001), increased productivity and N-fixation by N-fixers may also lead to reductions in wood density at sites where N was otherwise limiting.

5.6 WHAT ARE THE IMPACTS OF CLIMATE CHANGE ON FUTURE MANAGEMENT OF FARM AND FORESTRY SYSTEMS, AND LIKELY ADAPTATIONS, INCLUDING FERTILISER MANAGEMENT?

The main effects of climate change on agricultural and forestry systems are likely to be:

- Increased growth arising from eCO_2 and increased temperature if *other factors do not become limiting*. These other factors could be water, nutrients, pests or diseases and extreme weather events.
- Potential for more legume growth in mixed sward pastures and N-fixing shrubweed growth in forests – again if other factors do not become limiting.
- More winter growth (mild winters) …
- … but some of this might be offset by increased periods of summer drought
- More extreme storm events increasing the risk of soil erosion, particularly in hill country regions, and flooding, both of which will impact on NPP.
- Increase in frequency of hot days, which may impact on animal performance in pastoral systems.

Whilst there will be subtle changes to processes and transformation rates, the effects decribed below are probably the main high level effects.

Pastoral sector

The main limitations that may occur are:

- Progressive N limitation, where N from legumes cannot meet the shortfalls,
- Increased risk of drought reducing soil N inputs,
- Increased pest and disease pressure reducing soil N inputs,
- Increased temperature affecting animal performance which could lead to both positive and negative impacts on N cycling.

Based on the above, a likely response is:

- A focus on ensuring that factors do not limit yield potential e.g. greater N fertiliser use, where PNL is limiting pasture production,
- And/or increased legume use,
- Increasing use of irrigation, especially in the drier east and possibly north,
- Developing feed strategies to meet shortfalls due to increased summer drought frequency e.g. more on-farm supplement production, or increase in purchased feeds,
- Utilising the extra pasture that is grown through increased stocking rates,
- Increased steps to minimise adverse effects of hot days on animal performance.

Arable sector

The main limitations that may occur are:

- Reduced yield potentials and crop suitability,
- Increased risk of drought reducing soil N inputs,
- Increased pest and disease pressure reducing soil N inputs,
- Increased crop damage risk from extreme weather events.

Based on the above, a likely response is:

- A focus on ensuring that factors do not limit yield potential e.g. greater N fertiliser use,
- Increasing use of irrigation, especially in the drier east and possibly north,
- Reduced tillage to conserve soil moisture,
- Increasing use of mulches and other soil water conservation methods,
- Modifying crop selection and rotation to respond to opportunities and risks, including development of new or existing overseas markets.

Forestry sector

The main limitations that may occur are:

- Progressive N limitation, where soil N or N from under-storey legumes cannot meet the shortfalls,
- Increased risk of drought and fire leading to reduced soil N inputs,
- Increased pest and disease pressure leading to reduced soil N inputs,
- Increased competition from N-fixing shrubweeds.

Based on the above, a likely response is:

- Increased monitoring of soil and plant tissue to ensure potential growth responses to eCO_2 are not nutrient limited,
- Greater N fertiliser use, where PNL is limiting forestry production,
- Improved fire surveillance and adoption of measures to reduce drought impacts,
- Selection for nutrient uptake efficiency, and drought and disease tolerance,
- Increased use of herbicides to control N-fixing shrubweeds, gorse and broom.

These management responses do not factor in any changes to policy/regulation, e.g. restrictions on N fertiliser use or irrigation abstraction.

Identification of the knowledge and opportunity gaps for future work

To make progress, we need to:

- Understand impacts of individual elements of climate change on N processes (semi-quantitatively),
- Improve our understanding of the net effects of climate change at both a farm-scale and at individual processes level (e.g. through modelling),
- Understand the drivers of net directional change, since these will depend on individual circumstances,
- Have sufficient detailed understanding to aid modelling,
- Anticipate management response to climate change since this adds another level of complexity to assessing climate change effects at the system scale.

Whilst this literature review demonstrates that good progress has been made in many of these aspects, there are still gaps in knowledge:

Dissolved Organic Nitrogen (DON) – there is accumulating evidence that this form of soluble N can contribute a significant proportion of leachable N, yet little is known about this in the context of climate change. It could be hypothesised that this will become an increasingly important loss pathway for soluble N in a high C world.

Urine patch dynamics under climate change – urine patches are the engine room of N cycling in pastoral systems. More information is needed on climate change response of urine patch dynamics. Scenarios could be hypothesised where leaching or denitrification losses decrease (more pasture growth and N removal) or increase (summer drought limiting N removal). There is also a need to understand detailed processes (immobilisation, MIT). The impacts of climate change on urinary N source (pasture species and dry matter intakes) would also need to be quantified.

Responses in a 'nitrogen-rich' pastoral system – much of the research has been undertaken in low N inputs systems. There is an indication that PNL becomes less important in systems with more N. This needs to be tested, as does the systems response. We are extrapolating much of the information to dairy systems, whereas more research is needed in dairy systems. This is to confirm the response of dairy N cycling to climate change and to assess likely responses to climate change.

Implications of summer drought on overall response of the system – with the scenario of an increased likelihood of drought, then we need to better understand the implications of this on farm and forest systems and on N transformations. One possibility would be to use the 2008 and 2013 droughts as case studies.

Long-term dynamics and feedback effects - understanding the longer-term dynamics between PNL and grass and clover populations, and PNL and forest production and ecosystem N, is central to understanding climate change impacts on pasture and forestry systems.

Predicting farm responses to climate change - cropping systems models are used to project yield responses to climatic drivers at field scale. The extrapolation of plant temperature and CO_2 responses to much larger scales (e.g. ecosystem) is uncertain, with the science for linking models and data at different scales (plant, crop, paddock, farm, regional and global) in integrated assessments still in the early stages of development.

Predicting the interactive effects of plant water use, crop and soil management and climate on soil water conditions driving N processes - plants will become more water use efficient under eCO_2 conditions, producing more biomass per unit of water. However it is difficult to predict if future water use will increase, remain the same, or decline compared to now. In cropping systems actual water use will also depend on other factors including changes in evaporative losses, cropping cycles and changes in rotation. Improved understanding of how these interactions impact on soil water is required to better assess climate change impacts on N processes and losses.

Role of subsoil, soil and plant processes for regulating N losses - the impact on N losses of increasing labile C inputs through rhizosphere processes in subsoils is poorly understood or quantified. Increased C inputs via root exudation could stimulate denitrification, which on one hand could reduce potential nitrate leaching losses, but on the other could lead to increased N_2O emissions.

5.7 REFERENCES

Allard, V., Newton, P. C. D., Lieffering, M., Clark, H., Matthew, C., Soussana, J. F., Gray, Y. S. (2003). Nitrogen cycling in grazed pastures at elevated CO_2: N returns by ruminants. *Global Change Biology9*, 1731-1742.

Arnone Iii, J. A., Bohlen, P. J. (1998). Stimulated N_2O flux from intact grassland monoliths after two growing seasons under elevated atmospheric CO_2. *Oecologia 116*, 331-335.

Atiwill, P., Adams, M. (1993). Nutrient cycling in forests. *New Phytologist 124*, 561-582.

Baggs, E. M., Richter, M., Cadisch, G., Hartwig, U. A. (2003). Denitrification in grass swards is increased under elevated atmospheric CO_2. *Soil Biology and Biochemistry 35*, 729-732.

Barnard, R., Leadley, P. (2005). Global change, nitrification, and denitrification: A review. *Global Biogeochemical Cycles 19*, GB1007., doi:10.1029/2004GB002282.

Bazzaz, F., Miao, S. (1993). Successional status, seed size and responses of tree seedlings to CO_2, light and nutrients. *Ecology 74*, 104-112.

Beets, P., Gilchrist. K., Jeffreys, M. (2001). Wood density of radiata pine: effect of nitrogen supply. *Forest Ecology and Management 145*, 173-180.

Binkley, D., Burnham, H., Allen, H. (1999). Water impacts of forest fertilisation with nitrogen and phosphorus. *Forest Ecology and Management 121*, 191-213.

Borgesen, C. D., Olesen, J. E. (2011). A probabilistic assessment of climate change impacts on yield and nitrogen leaching from winter wheat in Denmark. *Natural Hazards and Earth System Sciences 11*, 2541-2553.

Brown, J. R., Blankinship, J. C., Niboyet, A., van Groenigen, K. J., Dijkstra, P., Le Roux, X., Leadley, P. W., Hungate, B. A. (2012). Effects of multiple global change treatments on soil N_2O fluxes. *Biogeochemistry 109*, 85-100.

Butler, J., Melillo, J., Johnson, J., Mohan, J., Steudler, P., Lux, H., Burrows, E., Smith, R., Vario, C., Scott, L., Hill, T., Aponte, N., Bowles, F. (2012). Soil warming alters nitrogen cycling in a New England forest: implications for ecosystem function and structure. *Oecologi 168*, 819-828.

Butterbach-Bahl, K., Dannenmann, M. (2011). Denitrification and associated soil N_2O emissions due to agricultural activities in a changing climate. *Current Opinion in Environmental Sustainability 3*, 389-395.

Campbell, B. D., Mckeon, G. M., Glifford, R. M., Clark, H., Stafford Smith, M. S., Newton, P. C. D., Lutze, J. L. (1996). Impacts of atmospheric composition and climate change on temperate and tropical pastoral agriculture. In: *Greenhouse : Coping With Climate Change* Bouma, W. J., Pearman, G. I., Manning, M. R. (Eds.) CSIRO Publishing Collingwood, Victoria, Australia, pp. 171-189.

Ceulemans, R., Janssens, I., Jach, M. (1999). Effects of CO_2 enrichment on trees and forests: Lessons to be learned in view of future ecosystem studies. *Annals of Botany 84*, 577-590.

Cheng, W. X., Johnson, D. W. (1998). Elevated CO_2, rhizosphere processes, and soil organic matter decomposition. *Plant and Soil 202*, 167-174.

Davis, M., Coker, G., Watt, M., Graham, D., Pearce, S., Dando, J. (2012). Nitrogen leaching after fertilising young Pinus radiata plantations in New Zealand. *Forest Ecology and Management 280*, 20-30.

de Graaff, M. A., Six, J., van Kessel, C. (2007). Elevated CO_2 increases nitrogen rhizodeposition and microbial immobilization of root-derived nitrogen. *New Phytologist 173*, 778-786.

Dieleman, W. I. J., Vicca, S., Dijkstra, F. A., Hagedorn, F., Hovenden, M. J., Larsen, K. S., Morgan, J. A., Volder, A., Beier, C., Dukes, J. S., King, J., Leuzinger, S., Linder, S., Luo, Y., Oren, R., De Angelis, P., Tingey, D., Hoosbeek, M. R., Janssens, I. A. (2012). Simple additive effects are rare: A quantitative review of plant biomass and soil process responses to combined manipulations of CO_2 and temperature. *Global Change Biology 18*, 2681-2693.

Dijkstra, F. A., Prior, S. A., Runion, G. B., Torbert, H.A., Tian, H., Lu, C., Venterea, R. T. (2012). Effects of elevated carbon dioxide and increased temperature on methane and nitrous oxide fluxes: evidence from field experiments. *Frontiers in Ecology and the Environment 10*, 520-527.

Dodd, M. B., Newton, P. C. D., Lieffering, M., Luo, D. (2010). The responses of three C4 grasses to elevated temperature and CO_2 in the field. *Proceedings of the New Zealand Grassland Association 72*, 61-66.

Drake, J., Gallett-Budynek, A., Hofmockel, K., Bernhardt, E., Billings, S., Jackson, R., Johnsen, K., Lichter, J., McCarthy, H., McCormack, M., Moore, D., Oren, R., Palmroth, S., Phillips, R., Pippen, J., Pritchard, S., Treseder, K., Schlesinger, W., DeLucia, E., Finzi, A. (2011). Increases in the flux of carbon belowground stimulate nitrogen uptake and sustain the long-term enhancement of forest productivity under elevated CO_2. *Ecology Letters 14*, 349-357.

Dueri, S., Calanca, P. L., Fuhrer, J. (2007). Climate change affects farm nitrogen loss - A Swiss case study with a dynamic farm model. *Agricultural Systems 93*, 191-214.

Elliot, A., Alexander, R., Schartz, G., Shanker, U., Sukias, J., McBride, G. (2005). Estimation of nutrient sources and transport for New Zealand using the hybrid mechanistic-statistical model SPARROW. *Journal of Hydrology (NZ) 44*, 1-27.

Fischer, G. (2009). World Food and Agriculture to 2030/50: How do climate change and bioenergy alter the long-term outlook for food, agriculture and resource availability? Expert Meeting on How to Feed the World in 2050. Food and Agriculture Organization of the United Nations. Economic and Social Development Department. Rome, Italy.

Fischer, G., Nachtergaele, F., Prieler, S., van Velthuizen, H. T., Verelst, L., Wiberg, D. (2008). Global Agro-ecological Zones Assessment (GAEZ 2007). IIASA, Laxenburg, Austria and FAO, Rome, Italy.

Garten, C., Iversen, C., Norby, R. (2011). Litterfall 15N abundance indicates declining soil nitrogen in free-air CO_2 enrichment experiment. *Ecology, 92*, 133-139.

Ghani, A., Dexter, M., Carran, R. A. (2007). Dissolved organic nitrogen and carbon in pastoral soils: the New Zealand experience. *European Journal of Soil Science 58*, 832-843.

Ghani, A., Muller, K., Dodd, M.,, Mackay, A. (2010). Dissolved organic matter leaching in some contrasting New Zealand soils. *European Journal of Soil Science 61*, 1512-1521.

Hartmann, A. A., Niklaus, P. A. (2012). Effects of simulated drought and nitrogen fertilizer on plant productivity and nitrous oxide (N_2O) emissions of two pastures. *Plant and Soil 361*, 411-426.

Hu, S., Chapin, F., Firestone, M., Field, C., Chiariello, N. (2001). Nitrogen limitation of microbial decomposition in a grassland under elevated CO_2. *Nature 409*, 188-191.

Hu, S., Tu, C., Chen, X., Gruver, J. B. (2006). Progressive N limitation of plant response to elevated CO_2: a microbiological perspective. *Plant and Soil 289*, 47-58.

Hu, S., Zhang, W. (2004). Impact of global change on biological processes in soil: Implications for agroecosystem management. *Journal of Crop Improvement 12*, 289-314.

Hungate, B. A., Dukes, J. S., Shaw, M. R., Luo, Y., Field, C. B. (2003). Nitrogen and Climate Change. *Science 302*, 1512-1513.

Kirschbaum, M., Watt, M., Tait, A., Ausseil, A-G. (2012). Future wood productivity of Pinus radiata in New Zealand under climate change. *Global Change Biology 18*, 1342-1356.

Klinedinst, P., Wilhite, D., Hahn, G. L., Hubbard, K. (1993). The potential effects of climate change on summer season dairy cattle milk production and reproduction. *Climatic Change 23*, 21-36.

Knoepp, J., Swank, W. (1995). Comparison of available soil nitrogen assays in control and burned forested sites. *Soil Science Society of America Journal 59*, 1750-1754.

Lam, S. K., Chen, D., Norton, R., Armstrong, R. (2013). The effect of elevated atmospheric carbon dioxide concentration on the contribution of residual legume and fertilizer nitrogen to a subsequent wheat crop. *Plant and Soil 364*, 81-91.

Lam, S. K., Chen, D. L., Norton, R., Armstrong, R., Mosier, A. R. (2012). Nitrogen dynamics in grain crop and legume pasture systems under elevated atmospheric carbon dioxide concentration: A meta-analysis. *Global Change Biology 18*, 2853-2859.

Larned, S., Scarsbrook, R., Snelder, T., Norton, N., Biggs, J. (2004). Water quality in low elevation streams and rivers of New Zealand: recent state and trends contrasting land-cover classes. *New Zealand Journal of Marine and Freshwater Research 38*, 347-366.

Larsen, K. S., Andresen, L. C., Beier, C., Jonasson, S., Albert, K. R., Ambus, P. E. R., Arndal, M. F., Carter, M. S., Christensen, S., Holmstrup, M., Ibrom, A., Kongstad, J., Van Der Linden, L., Maraldo, K., Michelsen, A., Mikkelsen, T. N., Pilegaard, K. I. M., PriemÉ, A., Ro-Poulsen, H., Schmidt, I. K., Selsted, M. B., Stevnbak, K. (2011). Reduced N cycling in response to elevated CO_2, warming, and drought in a Danish heathland: Synthesizing results of the CLIMAITE project after two years of treatments. *Global Change Biology 17*, 1884-1899.

Lenka, N. K., Lal, R. (2012). Soil-related Constraints to the Carbon Dioxide Fertilization Effect. *Critical Reviews in Plant Sciences 31*, 342-357.

Long, S. P., Ainsworth, E. A., Rogers, A., Ort, D. R. (2004). Rising atmospheric carbon dioxide: Plants FACE the Future. *Annual Review of Plant Biology 55*, 591-628.

Long, S., Ainsworth, E., Leakey, A., Morgan, P. (2005). Global food insecurity. Treatment of major food crops with elevated carbon dioxide or ozone under large-scale fully open-air conditions suggests recent models may have overestimated future yields. *Philosophical Transactions of the Royal Society B: Biological Sciences 360*, 2011-2020.

Luo, Y., Su, B., Currie, W. S. (2004). Progressive nitrogen limitation of ecosystem responses to rising atmospheric carbon dioxide. *BioScience 54*, 731-739.

Lu, M., Zhou, X, Yang, Q., Li, H., Luo, Y., Fang, C., Chen, J., Yang, X., Li, B. (2013). Responses of ecosystem carbon cycle to experimental warming: a meta-analysis. *Ecology 94*, 726-738.

McGroddy, M., Baisden, T., Hedin, L. (2008). Stoichiometry of hydrological C, N and P losses across climate and geology: an environmental matrix approachacross New Zealand primary forests. *Global Biogeochemical Cycles 22*, GB1026, doi:10.1029/2007GB003005.

Meisinger, J. J., Jokela, W. E. (2000). Ammonia volatilisation from dairy and poultry manure. In: *Managing nutrients and pathogens from animal agriculture* (NRAES-130), Natural Resource, Agriculture and Engineering Service, Ithaca, New York, pp. 334-354.

Melillo, J., McGuire, D., Kicklighter, D., Moore, B., Vorosmarty, C., Schloss, A. (1993). Global climate change and terrestrial net primary production. *Nature 363*, 234-240.

Mohan, J., Clark, J., Schleisinger, W. (2007). Long-term CO_2 enrichment of a forest ecosystem: implications for forest regeneration and succession. *Ecological Applications 17*, 1198-1212.

Newton, P. C. D., Lieffering, M., Bowatte, W. M. S. D., Brock, S. C., Hunt, C. L., Theobald, P. W., Ross, D. J. (2010). The rate of progression and stability of progressive nitrogen limitation at elevated atmospheric CO_2 in a grazed grassland over 11 years of Free Air CO_2 enrichment. *Plant and Soil 336*, 433-441.

Niklaus, P. A., Körner, C. (2004). Synthesis of a six-year study of calcareous grassland responses to in situ CO_2 enrichment. *Ecological Monographs 74*, 491-511.

Norby, R., Zak, D. (2011). Ecological lessons from free-air CO_2 enrichment (FACE) experiments. *Annual Review of Ecology, Evolution and Systematics 42*, 181-203.

Norby, R., Warren, J., Iversen, C., Medlyn, B., McMurtrie, R. (2010). CO_2 enhancement of forest productivity constrained by limited N availability. *Proceedings of the National Academy of Sciences 107*, 19368-19373.

Parfitt, R., Salt, G., Hill, L. (2002). Clear-cutting reduces nitrate leaching in a pine plantation of high natural N status. *Forest Ecology and Management 170*, 43-53.

Parfitt, R., Schipper, L., Baisden, W., Elliott, A. (2006). Nitrogen inputs and outputs for New Zealand in 2001 at national and regional scales. *Biogeochemistry 80*, 71-88.

Parry, M., Rosenzweig, C., Livermore, M. (2005). Climate change, global food supply and risk of hunger. *Philosophical Transactions of the Royal Society B: Biological Sciences 360*, 2125-2138.

Parsons, A. J., Thornley, J. H. M., Newton, P. C. D., Rasmussen, S., Rowarth, J. S. (2013). Soil carbon dynamics: The effects of nitrogen input, intake demand and off-take by animals. *Science of the Total Environment* (in press).

Phillips, R., Finzi, A., Bernhardt, S. (2011). Enhanced root exudation induces microbial feedbacks to N cycling in a pine forest under long-term CO_2 fumigation. *Ecology Letters 14*, 187-194.

Pleijel, H., Uddling, J. (2011). Yield vs. quality trade-offs for wheat in response to carbon dioxide and ozone. *Global Change Biology 18*, 596-605.

Prior, S. A., Runion, G. B., Marble, S. C., Rogers, H. H., Gilliam, C. H., Torbert, H. A. (2011). A Review of Elevated Atmospheric CO_2 Effects on Plant Growth and Water Relations: Implications for Horticulture. *HortScience 46*, 158-162.

Prior, S. A., Torbert, H. A., Runion, G. B., Rogers, H. H. (2004). Elevated atmospheric CO_2 in agroecosystems: Residue decomposition in the field. *Environmental Management 33,* S344-S354.

Pritchard, S. G. (2011). Soil organisms and global climate change. *Plant Pathology, 60*, 82-99.

Pritchard, S. G., Rogers, H. H. (2000). Spatial and temporal deployment of crop roots in CO_2-enriched environments. *New Phytologist, 147*, 55-71.

Quinn, J. M., Ritter, E. (2003). Effects of land use and pine forest logging on stream nutrients at Purukohukohu, Central North Island. *Proceedings of "Rotorua Lakes 2003: Practical Management for Good Lake Water Quality"*, Rotorua. pp149-157.

Reich, P. B., Hungate, B. A., Luo, Y. (2006). Carbon-nitrogen interactions in terrestrial ecosystems in response to rising atmospheric carbon dioxide. *Annual Review of Ecology, Evolution, and Systematics 37*, 611-636.

Rennenberg, H., Dannenmann, M., Gessler, A., Kreuzwieser, J., Simon, J., Papen, H. (2009). Nitrogen balance in forest soils: nutritional limitation of plants under climate change stresses. *Plant Biology 11 (Suppl 1)*,4-23.

Ross, D. J., Grayston, S. J., Whitehead, D. (2006). Changes in soil carbon and nitrogen properties and microbial communities in relation to growth of Pinus radiata and Nothofagus fusca trees after 6 years at ambient and elevated atmospheric CO_2. *Global Change Biology 12*, 1690-1706.

Ross, D., Newton, P., Tate, K. 2004. Elevated [CO_2] effects on herbage production and soil carbon and nitrogen pools and mineralization in a species-rich, grazed pasture on a seasonally dry sand. *Plant and Soil 260*, 183-196.

Saggar, S., Tate, K., Giltrap, D., Singh, J. (2008). Soil-atmosphere exchange of nitrous oxide and methane in New Zealand terrestrial ecosystems and their mitigation: a review. *Plant and Soil 309*, 25-42.

Schneider, M. K., Lüscher, A., Richter, M., Aeschlimann, U., Hartwig, U. A., Blum, H., Frossard, E., Nösberger, J. (2004). Ten years of free-air CO_2 enrichment altered the mobilization of N from soil in Lolium perenne L. swards. *Global Change Biology 10*, 1377-1388.

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

Silva, L., Anand, M., Leithad, M. (2010). Recent widespread tree growth decline despite increasing atmospheric CO_2. *PLos ONE 5*, e11543. Doi:10.1371/journal.pone.0011543.

Sjoeng, A. M. S., Kaste, O., Wright, R. F. (2009). Modelling future NO_3 leaching from an upland headwater catchment in SW Norway using the MAGIC model: II. Simulation of future nitrate leaching given scenarios of climate change and nitrogen deposition. *Hydrology Research 40*, 217-233.

Smith, K. (1997). The potential for feedback effects induced by global warming on emissions of nitrous oxide by soils. *Global Change Biology 3*, 327-338.

Smithwick, E., Kashian, D., Ryan, M., Turner, M. (2009). Long term nitrogen storage and soil nitrogen availability in post-fire lodgepole pine ecosystems. *Ecosystems 12*, 792-806.

Stuart, M. E., Gooddy, D. C., Bloomfield, J. P., Williams, A. T. (2011). A review of the impact of climate change on future nitrate concentrations in groundwater of the UK. *Science of the Total Environment 409*, 2859-2873.

Taub, D. R., Wang, X. (2008). Why are nitrogen concentrations in plant tissues lower under elevated CO_2? A critical examination of the hypotheses. *Journal of Integrative Plant Biology 50*, 1365-1374.

Teixeira, E. I., Brown, H. E. (2012). Climate change impacts on arable crop productivity. The New Zealand Institute for Plant & Food Research Limited, February 2012, SPTS No. 6607. Report prepared for the Foundation for Arable Research in New Zealand. 38 p.

Thomas, S. M., Cook, F. J., Whitehead, D., Adams, J.A. (2000). Seasonal soil-surface carbon fluxes from the root systems of young Pinus radiata trees growing at ambient and elevated CO_2 concentration. *Global Change Biology 6*, 393-406.

Thomas, S. M., Waterland, H., Dann, R., Close, M. E., Francis, G. S.,, Cook, F. J. (2012). Nitrous oxide dynamics in a deep soil-alluvial gravel vadose zone following nitrate leaching. *Soil Science Society of America Journal 76*, 1333-1346.

Torbert, H. A., Prior, S. A., Rogers, H. H., Schlesinger, W. H., Mullins, G. L., Runion, G. B. (1996). Elevated atmospheric carbon dioxide in agroecosystems affects groundwater quality. *Journal of Environmental Quality 25*, 720-726.

Tubiello, F. N., Soussana, J. F., Howden, S. M. (2007a). Crop and pasture response to climate change. *Proceedings of the National Academy of Sciences 104*, 19686-19690.

Tubiello, F. N., Amthor, J. S., Boote, K. J., Donatelli, M., Easterling, W., Fischer, G., Gifford, R. M., Howden, M., Reilly, J., Rosenzweig, C. (2007b). Crop response to elevated CO_2 and world food supply: A comment on "Food for Thought..." by Long et al., Science 312:1918-1921, 2006. *European Journal of Agronomy 26*, 215-223.

Turner, M. M., Henry, H. A. L. (2010). Net nitrogen mineralization and leaching in response to warming and nitrogen deposition in a temperate old field: The importance of winter temperature. *Oecologia 162*, 227-236.

van Groenigen, K. J., Osenberg, C. W.,, Hungate, B. A. (2011). Increased soil emissions of potent greenhouse gases under increased atmospheric CO_2. *Nature 475*, 214-U121.

van Kessel, C., Clough, T., van Groenigen, J. W. (2009). Dissolved Organic Nitrogen: An Overlooked Pathway of Nitrogen Loss from Agricultural Systems? *Journal of Environmental Quality 38*, 393-401.

Wachendorf, C., Taube, F., Wachendorf, M. (2005). Nitrogen Leaching from 15N Labelled Cow Urine and Dung Applied to Grassland on a Sandy Soil. *Nutrient Cycling in Agroecosystems 73*, 89-100.

Walthall, C. L., Hatfield, J., Backlund, P., Lengnick, L., Marshall, E., Walsh, M., Adkins, S., Aillery, M., Ainsworth, E. A., Ammann, C., Anderson, C. J., Bartomeus, I., Baumgard, L. H., Booker, F., Bradley, B., Blumenthal, D. M., Bunce, J., Burkey, K., Dabney, S. M., Delgado, J. A., Dukes, J., Funk, A., Garrett, K., Glenn, M., Grantz, D. A., Goodrich, D., Hu, S., Izaurralde, R. C., Jones, R. A. C., Leaky, A. D. B., Lewers, K., Mader, T. L., McClung, A., Morgan, J., Muth, D. J., Nearing, M., Oosterhuis, D. M., Ort, D., Parmesan, C., Pettigrew, W. T., Polley, W., Rader, R., Rice, C., Rivington, M., Rosskopf, E., Salas, W. A., Sollenberger, L. E., Srygley, R., Stöckle, C., Takle, E. S., Timlin, D., White, J. W., Winfree, R., Wright-Morton, L., Ziska, L. H. (2012). Climate Change and Agriculture in the United States: Effects and Adaptation. Washington, DC, pp. 186.

Watt, M., Whitehead, D., Richardson, B., Mason, E., Leckie, A. (2003). Modelling the influence of weed competition on the growth of young Pinus radiata at a dryland site. *Forest Ecology and Management 178*, 271-286.

West, J. W. (2003). Effects of Heat-Stress on Production in Dairy Cattle. *Journal of Dairy Science 86*, 2131-2144.

Xiang, S.-R., Doyle, A., Holden, P. A., Schimel, J. P. (2008). Drying and rewetting effects on C and N mineralization and microbial activity in surface and subsurface California grassland soils. *Soil Biology and Biochemistry 40*, 2281-2289.

Zac, D., Pregitzer, K., Kubiske, M., Burton, A. (2011). Forest productivity under elevated CO_2 and O_3: positive feedbacks to soil N cycling sustain decade-long net primary productivity enhancement by CO_2. *Ecology Letters 14*, 1220-1226.

Zak, D. R., Pregitzer, K. S., King, J. S., Holmes, W. E. 2000. Elevated atmospheric CO_2, fine roots and the response of soil microorganisms: a review and hypothesis. *New Phytologist 147*, 201-222.

Zanetti, S., Hartwig, U. A., Luscher, A., Hebeisen, T., Frehner, M., Fischer, B. U., Hendrey, G. R., Blum, H., Nosberger, J. (1996). Stimulation of symbiotic N_2 fixation in Trifolium repens L. under elevated atmospheric pCO_2 in a grassland ecosystem. *Plant Physiology112*, 575-583.

Zhang, W., Parker, K. M., Luo, Y., Wan, S., Wallace, L. L., Hu, S. 2005. Soil microbial responses to experimental warming and clipping in a tallgrass prairie. *Global Change Biology 11,* 266-277.

6 Knowledge gaps

Accurately predicting the possible impacts of climate-change on soil systems is difficult. Reasons for this include the inherent complexity of soil systems, the variable effects of climate change in different areas of New Zealand, and a lack of knowledge on key aspects of soil systems.

Four catergoreies of knowledge gaps in research have been identified and prioritised across the research topics. The highest priorities are marked with an H. The use of and development of robust long-term data sets and experiments are essential to addressing many of the knowledge gaps outlined below.

Soil natural capital

H Does stable soil carbon exist? What is the size of the carbon pool and under what time scales is this carbon stabilised? What is the temperature sensitivity of stabilised carbon?

H What is the spatial variability of soil carbon stocks? We can model them at a national scale, however, with large uncertainty. Models need to be refined to understand the spatial changes and related soil processes.

H What are the stocks and turnover rates of 'new' labile carbon, and 'old' protected carbon in New Zealand soils and can they be adequately represented in models to estimate national soil carbon stocks and change with climate change?

- What are the implications of soil type on nitrogen loss?
- What is the effect of increasing temperature on chemical stabilisation?
- What are the interrelationships between changes in vegetation and consequences for abundance and composition of the soil biotic community?
- Does climate change effect biota deep in the soil (> 30 cm depth) and if so does this have consequences for processes and services?
- How important are fungi in pastoral and cropping systems?

Soil cycling processes

H How does the composition of the soil biotic community affect soil processes, can we model these affects?

H How does spatial variability (e.g. urine patch dynamics on pasture; distribution of nitrogen in soil) influence nitrogen supply and losses under a changing climate?

H Can droughts and drought-breaking rain be better predicted so that we can understand their impact under contrasting land uses across different regions?

H What is the impact of climate change on nitrogen fixing plants that we are highly reliant on for nitrogen supply (pasture and legumes)?

- What are the processes responsible for dissolved organic nitrogen leaching, and how much is leached, as losses are likely to increase under a high carbon world?
- How can we improve our nutrient and carbon modelling to improve our understanding of nitrogen losses?
- What are the processes responsible for positively linking soil carbon with hydrophobicity (dry spots in paddock)?
- Does progressive nitrogen limitation occur under high nitrogen systems (e.g. cropping and intensive pastoral systems)?
- How does drought affect decomposition processes (via change in litter quality) and micro-nutrient availability?
- How can we better quantify interactions between temperature, moisture and carbon dioxide on carbon pathways of carbon supply to soils (as opposed to individual effects)?
- What are the impacts of increased climate variability on all relevant processes and carbon stocks?

Soil ecosystem services

H How can we predict ecosystem functional changes from soil carbon change to better understand the impact of climate change on soil ecosystem services?

H Changes in the soils ability to regulate pest/disease during climate change (beneficial organisms vs. pathogens/invaders) – this could be a major issue under climate change?

- What is the risk of net primary production under climate change being limited by a combination of nutrients?
- Does belowground (microbial) diversity affect regulating services?
- What is the effect of increasing temperature on chemical stabilisation?
- How does drought affect decomposition processes (via change in litter quality) and micro-nutrient availability?
- What are the processes responsible for positively linking soil carbon with hydrophobicity (dry spots in paddock)?

Land management

H What are the interactions of land management responses to climate change on the soil?
H Can land use specific models take into account management/plant species differences when addressing the impact of climate change on soil carbon?
H Are we managing for ecosystem function to provide economic and environmental benefits from soil?

- Will market drivers impact New Zealand’s soil more than climate change?
- What are the climate change impacts on trans-national carbon cycling (foot-printing e.g. life-cycle analysis)
- What alternate technologies e.g. biochar, could be used to regulate nitrogen processes and cycles and stock management?

Appendix 2 – Workshop participants

List of all the workshop participants at the soil and land use alliance (SLUA) workshop held in Wellington on the 25-26th of February 2013 to share, discuss and evaluate evidence for climate change impacts on soil processes affecting soil carbon stocks, soil nitrogen cycling and soil services, and what this means for forestry and farming in New Zealand.

Person	Company
Adrian Lill	Ministry for Primary Industries
Brent Clothier	Plant & Food
Brett Mullan	NIWA
Bryan Stevenson (Workshop Science co-ordinator)	Landcare Research
Carolyn Hedley	Landcare Research
Cecile de Klein	AgResearch
Dean Meason	Scion
Denis Curtin	Plant & Food
Edmar Teixeira	Plant & Food
Estelle Dominati	AgResearch
Frank Kelliher	Agresearch
Gavin Lear	University of Auckland
Gerald Rys	Ministry for Primary Industries
Ian Dickie	Landcare Research
Joanna Sharp	Plant & Food
Kate Parlane (Workshop organiser)	NZ Agricultural Geenhouse Gas Research Centre
Kate Orwin	Landcare Research
Kevin Tate	Landcare Research
Leo Condron	Lincoln University
Liz Keller	GNS
Loretta Garrett (Workshop Science co-ordinator)	Scion
Mark Shepherd	AgResearch
Michelle Rush (Facilitator)	Participatory Techniques Ltd
Mike Beare	Plant & Food
Mike Dodd	AgResearch
Mike Hedley	Massey University
Miko Kirschbaum	Landcare Research
Murray Davis	Scion
Nigel Bell	AgResearch
Paul Johnstone	Plant & Food
Paul Mudge	Landcare Research
Paul Newton	AgResearch
Pete Millard	Landcare Research
Peter Beets	Scion
Peter Kuikman	International
Roberta Gentile	Plant & Food
Simeon Smaill	Scion
Steve Thomas	Plant & Food
Steve Wakelin	AgResearch
Surinder Saggar	Landcare Research
Tim Clough	Lincoln University
Tim Payn	Scion
Tony van der Weerden (Workshop Science co-ordinator)	AgResearch
Troy Baisden	GNS
Val Snow	AgResearch

Appendix 3 – List of acronyms and abbreviations

AOB	Ammonia oxidising bacteria
AMF	Arbuscular mycorrhizal fungi
APSIM	Agricultural Production Systems sIMulator
BNF	Biological N Fixation
CenW	Carbon, Energy, Nutrients and Water
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CWT	Critical Water Threshold
DOC	Dissolved Organic Carbon
DON	Dissolved Organic Nitrogen
eCO_2	elevated atmospheric CO_2
EcM	Ectomycorrhizal fungi
EI	Ecoclimatic Index
DOC	Dissolved organic carbon
FACE	Free Air CO_2 Enrichment experiment
GCM	Global Climate Model
GHG	Greenhouse Gases- generally refers to CO_2, CH_4 and N_2O
IPCC	Intergovernmental Panel on Climate Change
IPO	Interdecadal Pacific Oscillation
MIT	Mineralisation Immobilisation Turnover
NIWA	National Instititue of Water and Atmospheric research
NPP	Net Primary Production
NZFACE	New Zealand Free Air CO_2 Enrichment (FACE) experiment
PET	Potential EvapoTranspiration
PNL	Progressive Nitrogen (or nutrient) Limitation
RCP	Representative Concentration Pathways
SLMACC	Sustainable Land Management and Climate Change
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
SOMD	Soil Organic Matter Decomposition
SRES	Special Report on Emissions Scenarios
SWR	Soil Water Repellency
TRFLP	Terminal Restriction Fragment Length Polymorphism
WUE	Water Use Efficiency