

# **Annual carbon balance of an intensively grazed pasture: magnitude and controls**

A thesis  
submitted in partial fulfilment  
of the requirements for the degree  
of  
**Master of Science in Earth and Ocean Sciences**  
at  
**The University of Waikato**  
by  
**PAUL MUDGE**

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# ABSTRACT

Soil carbon (C) is important because even small changes in soil C can affect atmospheric concentrations of CO<sub>2</sub>, which in turn can influence global climate. Adequate soil carbon is also required to maintain soil quality, which is important to if agricultural production is to be sustained. The soil carbon balance of New Zealand's pastoral soils is poorly understood, with recent research showing that soils under dairy pasture have lost large amounts of C during the past few decades.

The main objective of this research was to determine an annual farm scale C budget for an intensively grazed dairy farm, with a second objective being to determine the amount of CO<sub>2</sub>-C lost following cultivation for pasture renewal, and soil pugging by dairy cattle. A third objective was to investigate the environmental controls of CO<sub>2</sub> exchange in a dairy farm pasture system.

Net ecosystem exchange (NEE) of CO<sub>2</sub> was measured using an eddy covariance (EC) system from 15 December 2007 to 14 December 2008. Closed chamber techniques were used to measure CO<sub>2</sub> emissions from three cultivated paddocks and three adjacent pasture paddocks between 26 January 2008 and 5 March 2008. CO<sub>2</sub> emissions were also measured using chambers from pugged and control plots between 25 June and 5 August. Coincidentally this research was carried out in a year with a severe summer/autumn drought and a wetter than usual winter.

Annual NEE measured with the eddy covariance system was  $-1,843 \text{ kg C ha}^{-1}$  (a C gain by the land surface). Accounting for C in supplement import, milk export, pasture export and losses in methane, the dairy pasture system was a net sink of  $-880 \pm 500 \text{ kg C ha}^{-1}$ . This C sequestration occurred despite severe drought during the study, which was in contrast to other studies of grasslands during drought.

Cultivation under dry conditions did not increase cumulative CO<sub>2</sub>-C emissions compared to adjacent pasture paddocks. However, when C inputs to pasture paddocks via photosynthesis were included in calculations, net C loss from the cultivated paddocks (during the 39 day study) was estimated to be  $622 \text{ kg C ha}^{-1}$  more than the pasture paddocks. CO<sub>2</sub> emissions were lower from pugged plots compared to control plots, probably caused by decreased microbial and root respiration due to wetter soil conditions, and lowered root respiration as a result of lower pasture production.

Volumetric soil moisture content (soil moisture) had a dominant effect on CO<sub>2</sub> exchange at a range of temporal scales. Respiration and photosynthesis were both reduced when soil moisture was below 43% (~the lower limit of readily available water) and photosynthesis virtually ceased when soil moisture declined below 24% (~wilting point). Soil moisture also influenced the relationship between temperature and respiration and photosynthetic flux density (PPFD) and NEE.

These results suggest that management related soil disturbances of occasional cultivation for pasture renewal and soil pugging, are unlikely to cause large losses of soil C. Further, a severe drought also did not cause CO<sub>2</sub>-C losses from the land surface to the atmosphere on an annual scale, in contrast to previous studies.



# ACKNOWLEDGEMENTS

I would like to extend sincere thanks to the following people and organisations:

Louis Schipper my chief supervisor for all his help and for being so approachable. I really enjoyed our many interesting discussions, both on my thesis topic and in other areas not quite so related to it! Your enthusiasm was contagious.

Dave Campbell for his help with Matlab (the programme I've come to love to hate), advice on data analysis and his insightful comments on my thesis drafts. Your valuable advice and friendly manner have been greatly appreciated.

Craig Hosking, for help in the field, setting up databases and helping calibrate instruments. Jacinta Parenzee, for advice on laboratory things.

Tehani Kuske, for the work which she put into maintaining our shared field site.

Susanna Rutledge for your unfailing generosity in helping me time and time again with Matlab problems, and your invaluable encouragement.

Thank you to DairyNZ for allowing me to use Scott Farm as a field site. I am also very grateful for the assistance offered by the DairyNZ staff. Errol Thom, for being my point of contact at DairyNZ and taking the time to supply me with the information I required. A big thanks also to Cameron Clark and Gwyneth Verkerk for their assistance with (complicated) animal ethics approval forms. I am also grateful to John Siemelink and the other staff at Scott Farm, for keeping me well informed about the day to day management of the farm.

Thanks to the other Earth and Ocean Science Masters and PhD students, for their support, advice, encouragement and friendship.

Karsten Zegwaard for the useful discussions about pugging trial design.

DairyNZ and Landcare Research for their financial assistance with my research. I would also like to thank the sponsors of the following Scholarships, which I was very honoured to receive:

- The University of Waikato Masters Research Scholarship
- The Dr Stella Frances Scholarship (Environment Waikato and the Department of Conservation)
- The Todd Foundation Award for Excellence
- The Pukehou Poutu Scholarship
- The C. Alma Baker Postgraduate Scholarship

The financial support and encouragement which you gave is greatly appreciated.

Judith, for skewing my results (not really) by saving the worms in my bulk density cores. Thank you so much for all your love and support during this thesis, I could probably write another thesis if I was to include all the things I could thank you for...but I only have one line so thanks again and I love you very much.

Finally, God, who created such an amazing (and rather complicated!) universe.



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# 1. INTRODUCTION

## 1.1 Background

Soils are an important part of the global carbon cycle because they contain a large portion of the earth's actively cycling carbon (C). The top 1 m of soil is estimated to contain about 1500 Gt of C, which is about twice as much as in biomass (610 Gt) and the atmosphere (760 Gt) (Amundson, 2001). During the last few decades, concern about the rising concentration of CO<sub>2</sub> in the atmosphere and the potential implications on global climate has led to increased interest in understanding what drives changes in soil C. This has occurred because it is recognized that even small changes in soil C content could have a large effect on the global C budget (Tate *et al.*, 1997; Amundson, 2001). Carbon is also an important component of soil quality, through its influence on a range of soil properties and processes, such as; soil structure and stability, water holding capacity, cation exchange capacity and nutrient cycling (McLauchlan, 2006).

Changes in land use are well known to cause changes in soil C content and there has been considerable research conducted in this area (Guo and Gifford, 2002; Murty *et al.*, 2002). For example, conversion of forest or pasture to long term cultivation generally causes large losses of soil C (Davidson and Ackerman, 1993; Haynes and Tregurtha, 1999; Shepherd *et al.*, 2001), while soil usually gains C when cultivated land is converted back to pasture (Shepherd *et al.*, 2001; Guo and Gifford, 2002). Although it is clear that large changes in land use can alter soil C balances, the effects of changing management practices without obviously changing land use are not so well understood. About 26% of the earth's ice free land area is grazing land, with much of this land area under rapidly intensifying production systems (Steinfeld *et al.*, 2006). The large expanse of grazing land means the soil C balance of this land could have a large influence on the global C budget. Maintaining adequate C is also important if pasture productivity is to be maintained (Conant *et al.*, 2001).

Over half of New Zealand's land area is in agricultural production, with pastoral agriculture being the dominant land use (Parliamentary Commissioner for the Environment, 2004). Pastoral land is important to the economy of New Zealand, with about 40% of New Zealand's export earnings coming from products derived from pastoral agriculture, dairy products being the largest contributor (Parliamentary Commissioner for the Environment, 2004). Since its inception in New Zealand (~150 years ago) pastoral agriculture has continued to intensify, (MacLeod and Moller, 2006), in line with broader global trends (Steinfeld *et al.*, 2006). In recent decades, dairying has rapidly intensified and expanded in area, driven largely by high economic returns. For example, between 1992 and 2007 milk solid production (milk fat + milk protein) on New Zealand dairy farms increased by 43% (Livestock Improvement, 2007), due to increased use of nitrogen fertilizer, better pasture utilization, more supplementary feeding (Clark *et al.*, 2007) and higher stocking rates (Parliamentary Commissioner for the Environment, 2004). Intensification has also occurred on drystock farms.

The soil C balance of New Zealand's pastoral soils is poorly understood. Two early studies (Walker *et al.*, 1959; Jackman, 1964) demonstrated that following conversion of scrub or low producing pasture to improved pasture, C accumulated in the surface soil. Jackman (1964) also demonstrated that C in the surface soil reached a new equilibrium after about 7 – 40 years depending on soil type. Tate *et al.* (1997) re-sampled surface soils at 43 sites under pastoral agriculture that had been previously sampled 30 – 50 years earlier and found no systematic change in soil C. Subsequently it became generally accepted that soil C under established pasture in New Zealand was at near steady state (Saggar *et al.*, 2001a; Tate *et al.*, 2005). However, Schipper *et al.* (2007) demonstrated that during the 17 – 30 years prior to 2005, soils at 31 flat or gently rolling pastoral sites (mainly dairy farms) throughout New Zealand lost on average 1,060 kg C ha<sup>-1</sup> y<sup>-1</sup> (from the top ~1 m of soil). The cause(s) of the large C loss observed by Schipper *et al.* (2007) are not known.

Change in soil C content occurs when there is an imbalance between C inputs (photosynthesis or feed imports) and C outputs (respiration, product, methane, leaching and erosion) (Amundson, 2001). To date, most studies in New Zealand



investigating changes in soil C, have measured changes in soil C through time, which provides an integrated measure of all inputs and outputs. Another approach is to measure/calculate a farm scale C budget, accounting for all the C inputs and outputs from a farm system. In grazed pastures, pasture biomass is relatively constant at annual time scales and therefore any imbalance between C inputs and outputs is likely to be reflected in changes in soil C content. An advantage of the budgeting approach is that factors driving changes in soil C can be more easily identified (Ammann *et al.*, 2007).

Measuring the difference between C inputs via photosynthesis and outputs via respiration at the paddock scale is difficult, but can be achieved using the micrometeorological technique of eddy covariance (EC). EC systems continuously measure the exchange rate of CO<sub>2</sub> between the land surface and atmosphere, by measuring the covariance between vertical wind speed and CO<sub>2</sub> concentrations (Baldocchi, 2003). From these measurements, CO<sub>2</sub>-C budgets can be calculated for time scales ranging from hours to years. Most of the other C fluxes can be quantified reasonably accurately from farm records (e.g. milk and meat production, imported and exported supplementary feed). Losses of C in methane, leaching and erosion can be estimated based on literature values. In this way a full C budget for a pastoral system can be quantified. There has only been one such study on an intensively grazed pasture in New Zealand, and this was for a dairy farm on a drained peat soil (Nieveen *et al.*, 2005). There are also few international studies of intensively grazed pastures such as those common in New Zealand (Jaksic *et al.*, 2006; Byrne *et al.*, 2007). Extrapolation of data from Nieveen *et al.* (2005) to pastures on mineral soils would be highly questionable and further budgets for New Zealand dairy systems are needed.

Chamber measurements of CO<sub>2</sub> emissions from soil is another approach that can be used to quantify the effect that specific management practices have on soil C (Rochette and Hutchinson, 2005). The most well documented cause of C loss from soils is physical disturbance, which tends to destroy soil aggregates, exposing previously protected C to microbial decomposition (Tisdall and Oades, 1982; Six *et al.*, 2004; Grandy and Robertson, 2007). Two types of physical disturbance that occur in pastoral systems are cultivation, and pugging (the plastic

deformation of wet soil by animal hooves, Greenwood and McKenzie, 2001). Cultivation of pastoral soils is infrequent, generally only occurring about every 10 – 15 years on dairy farms, when pastures are re-sown and when crops are grown. However, the amount of C lost during these cultivation events and subsequent recovery is poorly quantified (Conant *et al.*, 2007). On intensive dairy farms in New Zealand, Singleton and Addison (1999) suggested that over a period of a few years pugging could occur over much of the farm. Although the effect of pugging on soil properties and pasture production has been well researched (Edmond, 1963; Menneer *et al.*, 2005b; Zegwaard, 2005), the effect of pugging on soil C has not been investigated.

## 1.2 Aims and objectives of this research

The overall aim of this research was to improve our understanding of the carbon balance of intensively grazed dairy farm systems.

The following objectives were set:

1. Determine an annual farm scale C budget for a dairy farm, by combining data from CO<sub>2</sub>-C exchange measurements made using an eddy covariance system, with other fluxes of C calculated from farm production data and literature values.
2. Quantify the CO<sub>2</sub>-C losses following cultivation for pasture renewal in late summer and soil pugging by dairy cattle in winter and compare these to losses from adjacent control plots and the annual C budget.
3. Investigate the environmental controls of CO<sub>2</sub> exchange in a dairy farm pasture system, including temperature, moisture and light.

By coincidence, there was a severe drought during the summer/autumn of 2008, with rainfall in January being the lowest in the last 100 years. In contrast, the winter months of July and August were much wetter than usual. These climatic extremes provided an ideal opportunity to investigate factors controlling CO<sub>2</sub> exchange and the impact on seasonal and annual C budgets.

### **1.3 Thesis layout**

Chapter 2 reviews literature on the soil carbon balance under pastoral agriculture and specifically investigates the effect that occasional cultivation and cattle pugging has on soil C in pastoral systems. The focus will be on New Zealand systems although international studies will be included where relevant.

Chapter 3 is the focus of the thesis and presents and discusses the annual C balance of a dairy farm system and the effect that cultivation and cattle pugging had on CO<sub>2</sub> emissions. Chapter 3 has been written in the form of a scientific paper and following submission of this thesis, will be condensed and submitted to a peer reviewed journal.

Chapter 4 describes how the environmental factors of temperature, soil moisture and light affected CO<sub>2</sub> exchange at Scott Farm at temporal scales ranging from minutes to annual trends.

A summary, conclusions and recommendations for further research are presented in chapter 5.

Because chapter 3 is written in the form of a paper there is some unavoidable overlap between the introduction in chapter 3 and the thesis introduction. In chapter 4 there is also some repetition of methods in chapter 3, to remind the reader of the general approach.



## **2. LITERATURE REVIEW**

### **2.1 Purpose and structure of this literature review**

This literature review will provide a synthesis of the current state of knowledge about the soil carbon balance under pastoral agriculture, with a focus on intensively grazed New Zealand pastoral systems. However, the review will begin with a brief explanation of the different forms of C found in soil and how C is stored in soil (section 2.2). A summary of the terminology commonly used in C budget accounting will then be given (section 2.3), followed by an outline of the main approaches used to measure changes in soil C (section 2.4). The magnitude of the key components (i.e. inputs and outputs) of the soil carbon balance of pastoral systems will then be presented (section 2.5). A review of studies into the soil carbon balance of pastoral agriculture follows, with sections on direct measurement of soil C and eddy covariance studies (section 2.6). This will be followed by an investigation into the effect that physical disturbance (cultivation and soil pugging) has on soil C and CO<sub>2</sub> emissions and the implications that these disturbances might have on the C balance of pastoral soils (section 2.7).

Each of these areas could form a literature review in their own right but the focus here is on annual scale fluxes of C in dairy systems (section 2.5 onwards).

### **2.2 Soil Carbon**

Carbon is present in soil in a number of different fractions, with differing turnover times. The most visually obvious C in soil is in living plant roots and soil organisms (earthworms and fungi) and un-decomposed plant and animal tissue. However, the majority of soil C is usually found in highly decomposed organic material, known as humus (Amundson, 2001). Humus is a mixture of carbon, nitrogen, phosphorus, oxygen, hydrogen and other elements. This combination of elements is commonly referred to as soil organic matter (SOM) which is about 55% carbon and 45% other elements (Blanco-Canqui and Lal, 2004).

SOM can be separated into physically and functionally distinct pools by physical fractionation techniques. Often SOM is described as being either ‘light fraction’ or ‘heavy fraction’ with these two fractions separated based on density (Collins *et al.*, 1997; Gregorich *et al.*, 2006; Tan *et al.*, 2007). Light fraction SOM is also known as uncomplexed SOM and is predominantly comprised of partially decomposed plant and animal materials that are not bound with sand, silt or clay particles (Christensen, 2001; Gregorich *et al.*, 2006). Uncomplexed SOM can be further divided into free or occluded SOM. Free SOM is comprised of loose organic material in the soil matrix (inter-aggregate), while occluded SOM is located within aggregates and is thus physically protected (Christensen, 2001). Light fraction SOM typically has residence times in the soil in the order of months to years (Post *et al.*, 2001) and can be strongly influenced by land use and management practices (Gregorich *et al.*, 2006; Tan *et al.*, 2007). Heavy fraction SOM is highly processed organic material that is strongly bound to the surfaces of primary sand, silt or clay particles (mainly clay) (Christensen, 2001). Heavy fraction SOM can have residence times of decades to centuries (Parton *et al.*, 1988; Jenkinson, 1990) and is less impacted by land management than light fraction SOM (Gregorich *et al.*, 2006).

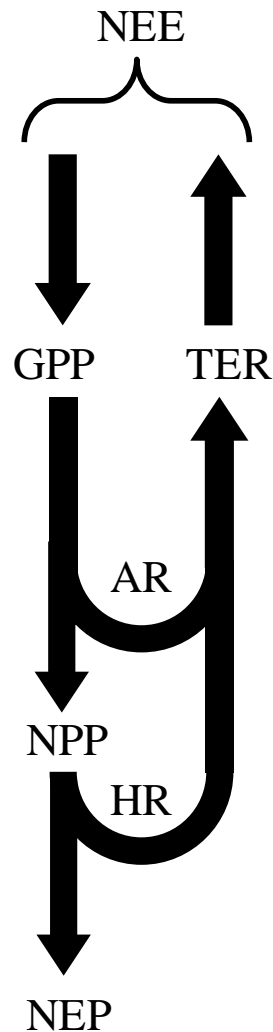
Aggregates have long been recognized to be important to soil function, and the complex relationship that exists between SOM and soil aggregate dynamics has received considerable research attention (Six *et al.*, 2004). The majority of SOM is usually found within aggregates and for most soils SOM plays a vital role in the formation and stabilization of aggregates (Tisdall and Oades, 1982; Oades and Waters, 1991). In turn, aggregates help protect SOM from microbial decomposition (Blanco-Canqui and Lal, 2004) and therefore facilitate carbon accumulation in soil. Factors that cause destruction of soil aggregates can lead to a loss of SOM because previously protected SOM becomes available for microbial decomposition (Collins *et al.*, 1997; Grandy and Robertson, 2007). Six *et al.* (2004) provide a comprehensive review of the complex relationship between aggregates and soil organic matter and also touches on other aspects affecting aggregate dynamics and the reader is referred to this review for further information.

The mineral constituents of a soil have been shown to affect both the amount of C a soil can store and the susceptibility of C to decomposition following disturbance. For example, in New Zealand, Allophanic Soils can contain up to twice as much C as other soil orders (Jackman, 1964; Percival *et al.*, 2000; Shepherd *et al.*, 2001), and the decline of C in Allophanic Soils following cultivation is much less than in other soils under the same treatments (Parfitt *et al.*, 1997; Shepherd *et al.*, 2001). The high C content of Allophanic Soils and their resistance to C loss under cultivation has generally been attributed to the presence of the clay mineral allophane, which has a very large surface area onto which organic molecules can be absorbed (Parfitt *et al.*, 1997). However, Percival *et al.* (2000) found a very poor correlation between allophane and C content in allophanic soils and suggested that the high C content of these soils may instead be related to the good physical properties and naturally high P contents which would have led to high primary production and thus high C inputs from the original native forests. Percival *et al.* (2000) demonstrated that pyrophosphate extractable aluminium was the best predictor of soil C content in New Zealand soils.

Stabilization of organic matter in soil is a large and continuous area of research, but not the focus of this study. Further reviews are provided by Christensen (2001), Blanco-Canqui and Lal (2004) and Six *et al.* (2004).

### **2.3 Terminology associated with ecosystem level C budgeting**

A range of terms are used to describe the stocks and flows of C within an ecosystem. This section will define some of these terms and acronyms commonly used in C budget accounting. A diagram showing the key components that contribute to CO<sub>2</sub> exchange between the land surface and atmosphere is presented in Fig. 2.1.



**Fig. 2.1.** Diagram showing the terms commonly used to describe CO<sub>2</sub> exchange between the land surface and atmosphere. GPP stands for gross primary productivity, TER, total ecosystem respiration, AR, autotrophic respiration, HR heterotrophic respiration, NPP, net primary production, NEP, net ecosystem production and NEE net ecosystem exchange (adapted from (Luyssaert *et al.*, 2007)).

Gross primary production (GPP) is the total amount of CO<sub>2</sub> fixed by plants via photosynthesis (Luyssaert *et al.*, 2007). Total ecosystem respiration (TER) is the respiration of all organisms (micro-organisms, plants, animals) within an ecosystem. TER can be further divided into autotrophic respiration (AR) and heterotrophic respiration (HR). AR respiration is defined in its broadest sense as the respiration of above and below ground parts of plants (Chapin *et al.*, 2006). However, it is difficult to distinguish between autotrophic root respiration and the respiration of microbes that are decomposing fresh root exudates and therefore the term rhizosphere respiration is commonly used to describe the contribution of plant roots to total soil respiration (Hanson *et al.*, 2000; Wan and Luo, 2003). HR respiration is respiration of all heterotrophic organisms, such as soil fauna and



animals (e.g. grazing cattle). Net primary production (NPP) is defined as GPP – AR (Chapin *et al.*, 2006). Most NPP is allocated to the production of plant biomass, with lesser amounts being exuded from roots, transferred to mycorrhizal fungi and lost as volatile organic compounds (Luyssaert *et al.*, 2007). Net ecosystem production (NEP) is GPP – TER, and is therefore the balance between total biological fixation of C and total biological respiration. Net ecosystem exchange (NEE) is the net exchange rate of CO<sub>2</sub> between the land surface and the atmosphere (Baldocchi, 2003). NEE can differ slightly from NEP because NEE not only includes the balance between the biological process –GPP and TER, but also non-biologically produced CO<sub>2</sub> fluxes associated with fire and ultraviolet oxidation of organic matter (Chapin *et al.*, 2006). NEP was a term developed by ecologists and therefore C inputs to ecosystems are defined as positive and losses negative. On the other hand, NEE is a term used by atmospheric scientists and has the opposite sign with C inputs to ecosystems defined as negative and losses to the atmosphere positive (Chapin *et al.*, 2006).

CO<sub>2</sub> is not the only pathway via which C can be gained or lost from an ecosystem and other pathways must also be accounted for when developing total C budgets. In pastoral ecosystems, there can be losses of C in product (e.g. meat and milk, plant biomass), in methane and via leaching and erosion. Inputs from supplementary feeding and manure can also occur. Chapin *et al.* (2006) proposed a new term to be used in carbon budget accounting for ecosystems—net ecosystem carbon balance (NECB), which is intended to include all C inputs and outputs. A simplified equation for NECB for a pastoral system can be written as follows:

$$\text{NECB} = -\text{NEE} + F_{\text{import}} - F_{\text{harvest}} - F_{\text{product}} - F_{\text{CH}_4} - F_{\text{leach}}$$

Where –NEE is NEE with the sign reversed,  $F_{\text{import}}$  is C brought into the system via manure or supplementary feed,  $F_{\text{harvest}}$  is C exported in harvested plant biomass,  $F_{\text{product}}$  is C exported in animal products (e.g. meat, milk, wool),  $F_{\text{CH}_4}$  is methane lost from grazing ruminants and manure (methane oxidation is not included since it is usually low in agricultural systems) and  $F_{\text{leach}}$  is C lost via leaching (as dissolved organic or inorganic C). NECB has the opposite sign to

NEE, with gains of C by the ecosystem being positive and losses from the ecosystem negative.

This reversal of sign between NEE and NECB could lead to confusion when trying to discuss NEE and NECB in close proximity to each other. The following two sentences are an example of the confusion this reversal of sign could cause. “*The annual NEE of  $-1,000 \text{ kg CO}_2\text{-C ha}^{-1}$  showed that the ecosystem was a net sink of  $\text{CO}_2\text{-C}$ . However, when all C inputs and outputs were accounted for, the sink was smaller with the NECB being only  $+500 \text{ kg C ha}^{-1}$ ”.* It seems logical in ecosystem studies, that the sign of either NEE or NECB be reversed to avoid this confusion. Soussana *et al.* (2007) used the following equation to calculate the NECB:

$$\text{NECB} = \text{NEE} + F_{\text{import}} + F_{\text{harvest}} + F_{\text{product}} + F_{\text{CH}_4} + F_{\text{leach}}$$

where a negative NECB denotes C uptake by the ecosystem and a positive NECB a C loss from the ecosystem, ( $F_{\text{import}}$  has a negative sign). This is the approach that will be used in this thesis.

## **2.4 Methods used to measure changes in soil carbon content**

While there has been considerable research into changes in soil C over the past few decades, there is no internationally agreed method to measure such changes (Post *et al.*, 2001). This section outlines some of the methods used to measure changes in soil C and discusses their advantages and disadvantages.

### **2.4.1 Direct measurements of soil carbon**

A number of approaches that involve direct measurement of soil C content have been used to measure how soil C changes with time. A common approach is to compare the C content of samples taken at one point in time with those taken from the same area at later point in time. An advantage to this approach is that measured changes in soil C are an integrated measure of many years of management. Although this sounds like a simple task, there are a number of factors that limit the accuracy of this re-sampling approach. For relatively short

time periods (e.g. <10 years) between samplings the magnitude of any C loss or gain from soil is usually small (generally 0 – 10,000 kg C ha<sup>-1</sup> y<sup>-1</sup>) in relation to the large background C content (e.g. NZ average is 168,000 kg C ha<sup>-1</sup> in the top 1 m, Tate *et al.*, 1997). In addition, spatial variation in soil C content is usually large (Brye *et al.*, 2002) and therefore detecting changes in soil C over short time periods can be difficult (Post *et al.*, 2001; Bossuyt *et al.*, 2002). Another difficulty is that soil cores taken for C analysis are usually taken to a set depth and therefore if changes in bulk density have occurred (e.g. due to cultivation) a loss/gain of C may be found even if there was in fact no change in C. This problem can be overcome by sampling (or correcting) to an equivalent mass of soil or sampling to a much greater depth which will reduce the effect of any changes in bulk density at the surface.

Methods for analysis of soil C have also changed in the last few decades from wet chemical oxidation (i.e. the Walkley-Black method) to dry combustion in a resistance or induction furnace (Collins *et al.*, 1997). Consequently, historical C values for soil using the Walkley-Black method may need to be calibrated against newer methods where historical soil samples have been archived.

Choosing an appropriate sampling depth relative to the specific research question is another important consideration when trying to determine if changes in land use has caused a change in soil C. Conservation tillage has become widely accepted to cause carbon sequestration because many studies have reported accumulation of C in the surface soil (usually 0-300 mm) compared with conventionally tilled soil (Baker *et al.*, 2007). However, Baker *et al.* (2007) suggested that the widespread belief that conservation tillage causes C sequestration “may simply be a artefact of sampling methodology”. Recently, Blanco-Canqui and Lal (2008) provided further supporting evidence for this suggestion by demonstrating that when conventionally tilled and no-tilled soils were sampled to 600 mm depth there was generally no difference in total C content.

Another approach used to try and understand how soil C changes with time is sampling from natural or man-made chronosequences of vegetation and/or land use on the same soil type (Jackman, 1964). This approach is essentially

substituting space for time (Baldocchi, 2008), which can allow for longer term investigations because studies can be carried out at sites where soil samples were not previously acquired. Sparling *et al.* (2003) validated this approach by re-sampling a chronosequence of recovering slip sites and showed that C accumulation rates determined from chronosequences were the same as sampling through time. A similar approach to chronosequences is to use paired site comparisons, where soils in close proximity, but under different land uses are sampled at one point in time (Sparling *et al.*, 2000). This method is based on the assumption that the original soil C content was the same under both land uses.

The major limitation of all direct sampling techniques is that generally long time periods are required to detect changes in soil C and such sampling yields little information on underlying processes that might be driving changes in soil C (Ammann *et al.*, 2007).

### **2.4.2 Measuring CO<sub>2</sub> emissions**

A more direct approach being increasingly utilized to understand changes and controlling factors of soil C, is the measurement of CO<sub>2</sub> fluxes to or from the soil using Infrared Gas Analysers (IRGA's) (Rochette and Hutchinson, 2005). IRGA's are extremely sensitive and can detect small changes in CO<sub>2</sub> concentrations over very short time periods (seconds). This sensitivity makes IRGA's ideally suited to isolating the effect of specific management events (e.g. grazing or cultivation) or environmental change (e.g. rainfall) on CO<sub>2</sub> exchange, from which changes in soil C can be calculated. IRGA's can be used in conjunction with chambers to measure CO<sub>2</sub> emissions from small areas (<1 m<sup>2</sup>), and are also a key component of the micrometeorological technique—eddy covariance which measures CO<sub>2</sub> exchange between the land surface and atmosphere at hectare scales. Both these techniques are discussed below.

#### **2.4.2.1 Chamber measurements**

Prior to 1985, most measurements of in situ soil respiration were made using the alkali trap method (Rochette and Hutchinson, 2005). In this method, a sealed chamber containing a known quantity of an alkali substance (usually NaOH) is inserted into the ground and typically left for 12-24 hours. CO<sub>2</sub> leaving the soil

surface is absorbed by the alkali and the quantity of CO<sub>2</sub> trapped is subsequently determined by titration in the laboratory. Another chamber method is the static chamber method. This method involves placing a sealed chamber of known volume over the soil and taking at least three air samples at known intervals after the chamber was sealed (Grandy and Robertson, 2006). These air samples are then taken back to the laboratory and the CO<sub>2</sub> concentration measured using an IRGA. The respiration rate is then calculated from the change in CO<sub>2</sub> concentration with time.

The advent of portable IRGA's has meant that the alkali trap and static chamber methods are now less common. There are two common chamber techniques that are used in conjunction with portable IRGA's to provide measurements of soil respiration in the field: dynamic chamber and closed chamber systems.

When using both the dynamic chamber and closed chamber techniques, collars are usually inserted into the ground, and then the chamber placed on these collars. This ensures that there is a seal between the soil surface and the chamber. The dynamic chamber system works by passing a constant rate of air from outside the chamber through the chamber to ensure that the concentration of CO<sub>2</sub> (and water vapour etc) in the chamber remains similar to ambient conditions outside the chamber. This ensures that CO<sub>2</sub> emissions are not constrained by CO<sub>2</sub> build up in the chamber. The respiration rate is calculated from the difference between the CO<sub>2</sub> concentration in the air entering the chamber and the air leaving the chamber (Rochette and Hutchinson, 2005). This approach is ideal for making continuous measurements of soil respiration.

With the closed chamber technique, air within the collar and chamber is completely isolated from the atmosphere. Air is circulated from the chamber through the IRGA, which measures the temporal change in CO<sub>2</sub> concentration to calculate the CO<sub>2</sub> flux. One of the problems with this technique is that the concentration of CO<sub>2</sub> in the chamber increases during a measurement, which causes the rate of CO<sub>2</sub> efflux to decrease because the concentration gradient between the soil and the atmosphere decreases (Luo and Zhou, 2006). To minimize these errors fluxes are measured over short time periods and algorithms

are used to predict the respiration rate at the time the chamber closed (Luo and Zhou, 2006). Another approach to minimize CO<sub>2</sub> accumulation is to use chemicals that reduce the CO<sub>2</sub> concentration in the chamber to slightly below ambient so that the subsequent measurement is made as the concentration builds from below ambient to slightly above ambient.

Chambers can be used to make spot measurements or on a continuous basis. Spot measurements are made by using portable survey chambers and are ideal for determining spatial variation of soil respiration. On the other hand, automatic chambers are ideal for determining diurnal trends and the effect of specific environmental events such as rainfall on CO<sub>2</sub> emissions. High costs of automatic chambers often constrains their use only one or a few sites.

Using chambers is ideal for small plot trials investigating the effect that specific experimental manipulations have on CO<sub>2</sub> emissions. However, one major problem when using the chamber approach is that studies tend to be labour intensive, with multiple samples in space and time required to overcome problems of spatial and temporal variability (Eriksen and Jensen, 2001; Luo and Zhou, 2006).

#### ***2.4.2.2 The eddy covariance technique***

Eddy covariance (EC) is a micrometeorological technique commonly used to study CO<sub>2</sub> exchange between the land and atmosphere. CO<sub>2</sub> is transported between the land surface and the atmosphere by wind eddies. EC systems make rapid measurements (10 – 20 Hz) of CO<sub>2</sub> concentrations and windspeed in these eddies. The net exchange rate of CO<sub>2</sub> between the land and atmosphere (NEE) is calculated from the covariance between vertical wind speed and CO<sub>2</sub> concentrations (Baldocchi, 2003). CO<sub>2</sub> exchange rate is usually averaged over 30 minute periods and provides an almost continuous measure of NEE over areas of hectares in an upwind direction from the sensors. This technique overcomes many of the problems of spatial and temporal variability experienced when using chamber techniques. Near continuous measurements also presents opportunities to identify specific processes that affect CO<sub>2</sub> exchange (Ammann et al., 2007). The large scale approach of EC makes it ideally suited to measurement at the paddock

scale on farms. EC is now a common technique that has been widely used, with more than 400 long term sites situated all around the world (Baldocchi, 2008).

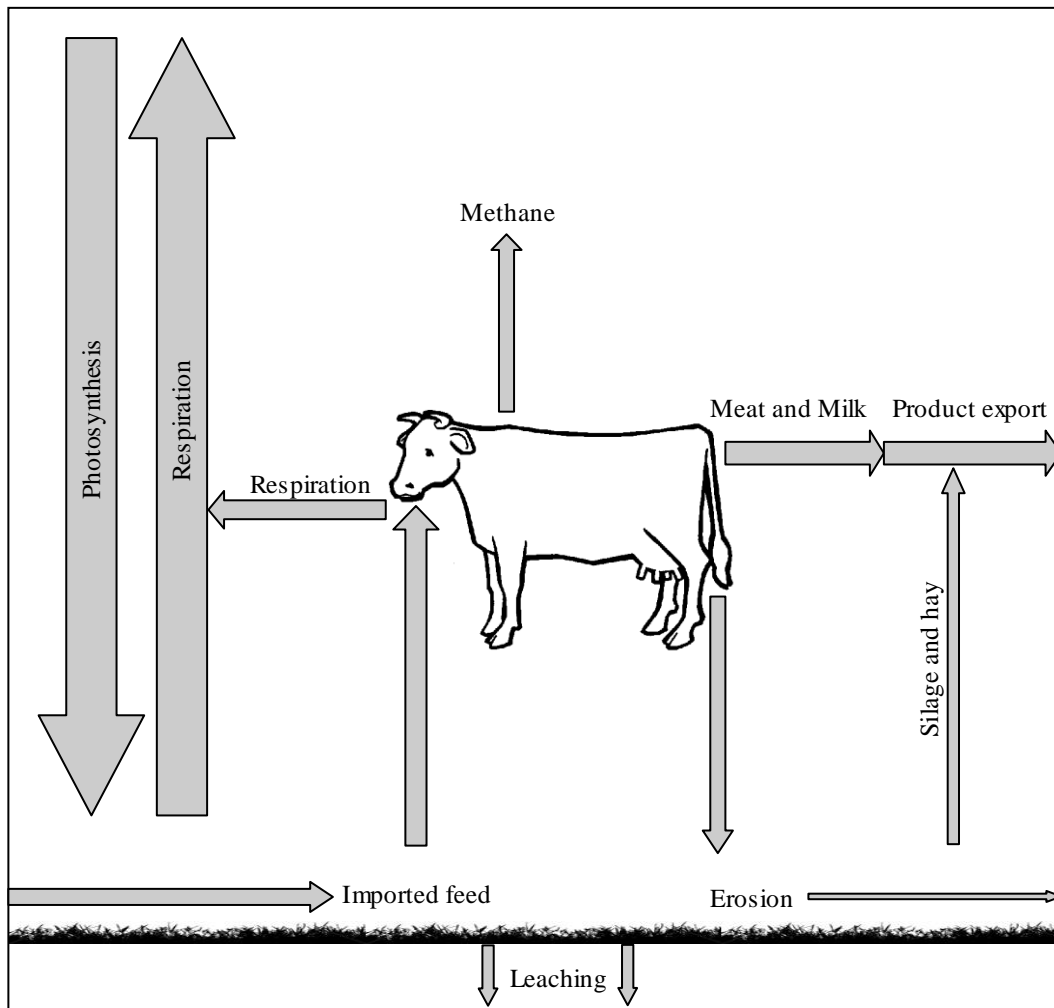
There are, however, many limitations to the EC technique. One of the major limitations is that to work optimally, EC systems require flat terrain and adequate fetch. Fetch is the distance over the surface of interest and adequate fetch is roughly 100 times greater than the height of the sensors (typically 2 – 10 m over grasslands). It has also been well documented that when windspeeds are low at night, NEE can be significantly underestimated because CO<sub>2</sub> accumulates (or drains away on sloping terrain) under the sensors. Sensors can also be affected by rainfall or just simply fail. These issues mean that in any study there are always gaps in the data (typically about 65 %, Falge *et al.*, 2001). If the aim of the study is to calculate daily, monthly or annual budgets these gaps must be filled. Because gaps are not randomly distributed, filling these gaps can produce biases. Another drawback of the EC approach is that EC systems are expensive and a high degree of technical expertise is required for appropriate installation maintenance and data corrections and analysis. The expense of EC systems and high operating expertise, generally means that there is little or no replication in ecosystem studies and errors are poorly known for specific sites. However, the accuracy of annual sums of NEE measured using the EC technique typically range from 300 to 1000 kg ha<sup>-1</sup> y<sup>-1</sup> (Baldocchi, 2008) with accuracy strongly influenced by site conditions (Baldocchi, 2003).

While the EC technique does not directly measure gross primary production (GPP) or total ecosystem respiration (TER) these values can be calculated from NEE data, providing valuable information to help improve our understanding of factors controlling NEE (Reichstein *et al.*, 2005; Gilmanov *et al.*, 2007).

## **2.5 Components of the soil carbon balance under grazed pasture**

The C content of pastoral soils is determined by the balance between C inputs (photosynthesis, imported feed and manure) and C outputs (respiration, leaching, product export, methane and erosion) (Amundson, 2001) (Fig. 2.2). This section

will outline the approximate magnitude of these inputs and outputs of C from pastoral soils.



**Fig. 2.2.** Schematic diagram of the key components that affect the C balance of a dairy farm soil. The size of the arrows is a very approximate indication of the size of the different fluxes.

## 2.5.1 Inputs

### 2.5.1.1 On site photosynthesis

On-site photosynthesis is usually the major pathway of C input to pastoral soils (Byrne and Kiely, 2006; Soussana *et al.*, 2007). C fixed by pasture plants can enter the soil via a number of different pathways. Above-ground plant material can enter the soil surface as litter or in dung and urine from grazing animals, while dead roots and root exudates enter the soil via the subsurface. The majority of C inputs to the soil are generally via subsurface root exudates and root death and decay (Saggar and Hedley, 2001).



In New Zealand, high producing intensively grazed dairy farm pastures typically produce about 15,000 kg of above ground (harvestable) dry matter (DM)  $\text{ha}^{-1} \text{y}^{-1}$  (Clark *et al.*, 2007) of which about 40% is C (Saggar and Hedley, 2001). Using  $^{14}\text{C}$  pulse labelling techniques Saggar and Hedley (2001) calculated that root production in an intensively grazed dairy farm pasture in New Zealand was almost exactly the same as above-ground shoot production. Therefore net primary production (NPP) on dairy farms would be about 30,000 kg DM  $\text{ha}^{-1} \text{y}^{-1}$ , equivalent to 12,000 kg C  $\text{ha}^{-1} \text{y}^{-1}$ . NPP, however, is not the total amount of C fixed by plants which is called gross primary production (GPP). GPP is NPP plus autotrophic respiration (AR). AR of perennial ryegrass pastures is about 45% of total respiration (Kuzyakov, 2002; Byrne and Kiely, 2006) and assuming that total respiration is the same as GPP (Baldocchi, 2008), the total GPP of intensively grazed New Zealand pastures would be about 17,400 kg C  $\text{ha}^{-1} \text{y}^{-1}$  (12,000 + 5,400 kg C  $\text{ha}^{-1} \text{y}^{-1}$ ). The numerous assumptions required to calculate GPP from NPP, clearly shows that more direct methods are needed to gain confidence in estimates of GPP. The EC technique is one such method that provides more direct measurement based estimates of GPP (Gilmanov *et al.*, 2007).

In one of the few eddy covariance studies over pasture in New Zealand, Nieveen *et al.* (2005) calculated GPP to be 13,486 kg C  $\text{ha}^{-1} \text{y}^{-1}$  (on a dairy farm on a drained peat soil in the Waikato). Other eddy covariance studies in grasslands around the world have found similar values for GPP (Table 2.1). Gilmanov *et al.* (2007) found that GPP at 20 European grassland sites ranged from 4,636 kg C  $\text{ha}^{-1} \text{y}^{-1}$  “in dry semi-natural pastures to 18,816 kg C  $\text{ha}^{-1} \text{y}^{-1}$  in intensively managed Atlantic grasslands” with the average over all 20 sites being 12,612 kg C  $\text{ha}^{-1} \text{y}^{-1}$ . On a dairy farm in Southern Ireland Jaksic *et al.* (2006) calculated GPP to be 16,730 kg  $\text{ha}^{-1} \text{y}^{-1}$  in 2002 and 17,180 kg  $\text{ha}^{-1} \text{y}^{-1}$  in 2003 and in an upland pasture in France, GPP was 15,940 kg  $\text{ha}^{-1} \text{y}^{-1}$  in an intensively grazed treatment and 15,140 kg  $\text{ha}^{-1} \text{y}^{-1}$  in an extensive treatment (Allard *et al.*, 2007).

### **2.5.1.2 Imported feed and manure**

Traditionally, New Zealand’s pastoral agriculture systems have been almost exclusively pasture based. Hay and silage grown on farms, to conserve the spring pasture surplus, was fed on the same farm during periods of feed deficit (e.g.

autumn/winter). As farming systems have intensified (particularly dairy farms) there has been an increasing trend to import supplementary feed from outside the farm boundaries (increasingly from overseas) to increase production and/or meet feed deficits caused by drought (MacLeod and Moller, 2006). During 2003-2004, 61% of dairy farms in New Zealand imported hay and silage from outside the farm boundaries, 24% imported maize silage and 37% imported grain or meal (Clark *et al.*, 2007). These feed imports would typically be about 10% of the total pasture consumed (C. Clark, pers. comm. 2008<sup>1</sup>), and total pasture consumption would be about 80% of total above ground pasture production ( $\sim 6,000 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ). Therefore, C import in supplements would be about  $500 \text{ kg ha}^{-1} \text{ y}^{-1}$  which is only 2.8 % of GPP. A small percentage of farms receive manure/effluent from piggeries and dairy factories which would be an additional input of C to the farm system.

## 2.5.2 Outputs

### 2.5.2.1 Respiration

Respiration is the largest loss of C from soil and on annual time scales is usually of similar magnitude to inputs via photosynthesis (Baldocchi, 2008) (Table 2.1). Soil respiration is usually divided into two components: heterotrophic respiration of SOM by soil organisms and rhizosphere respiration. Rhizosphere respiration is comprised of respiration by plant roots (autotrophic respiration) and heterotrophic respiration of soil fauna decomposing fresh root exudates (Hanson *et al.*, 2000; Wan and Luo, 2003). In pasture systems, rhizosphere respiration typically accounts for 20 – 50% of total soil respiration (Craine *et al.*, 1999; Kuzyakov, 2002; Wan and Luo, 2003; Byrne and Kiely, 2006).

Respiration of grazing animals is not a direct loss of C from the soil, but must be considered when calculating total carbon budgets for a soil/pasture system. About 70% of C consumed by grazing animals is respired as  $\text{CO}_2\text{-C}$ , with the remainder being partitioned between product (milk, meat or wool), dung and urine and methane (Jaksic *et al.*, 2006). Therefore on a dairy farm producing  $6,000 \text{ kg C ha}^{-1}$

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<sup>1</sup> Personal communication with Dr. Cameron Clark, Scientist, Dairy NZ, Hamilton.

$^1 \text{ y}^{-1}$  (15,000 kg DM  $\text{ha}^{-1} \text{ y}^{-1}$ ) of above-ground C of which 80% is consumed by cows (4,800 kg C  $\text{ha}^{-1} \text{ y}^{-1}$ ) would result in approximately 3,360 kg C  $\text{ha}^{-1} \text{ y}^{-1}$  respired to the atmosphere as  $\text{CO}_2\text{-C}$  from the animal. Increased pasture utilization would increase the amount of C lost via respiration (and product) leaving less to decompose and enter the soil as litter.

### 2.5.2.2 *Product*

Animal products, such as meat, milk and fibre and exports of plant material as silage or hay are another pathway via which C is lost from pastoral systems. For dairy farm systems the greatest C loss is usually in milk. In New Zealand, milk production is reported as milk solids (MS), which is the sum of milk fat + milk protein. Average milk solid production on New Zealand dairy farms for the 2006/2007 season was 934 kg MS  $\text{ha}^{-1} \text{ y}^{-1}$  (Livestock Improvement, 2007). Each kilogram of milk solids is equivalent to 0.834 kg of C when including lactose (Wells, 2001) and therefore the average C export in milk from dairy farms in New Zealand for the 2006/2007 season was 779 kg C  $\text{ha}^{-1}$ . Most dairy farms in New Zealand graze replacement heifers off farm (i.e. they are sent to another farm from birth until about two years of age). Therefore there would be little increase in live weight between heifers entering the herd (as two year olds) and older cows leaving as culls and so little C would be exported in meat.

C export in silage and hay is usually minimal on most New Zealand farms with most silage and hay being fed back on the same farm where it was grown. However, some dairy farm runoffs (where replacement heifers are grazed) and some sheep and beef farms export silage and hay, usually to dairy farms. The amount of C exported would generally not exceed  $\sim 2,500 \text{ kg C ha}^{-1} \text{ y}^{-1}$  (based on two harvests of 3,000 kg DM  $\text{ha}^{-1}$  and 40% C), from the specific paddocks from which the pasture was harvested, but generally only be a small proportion of the whole farm would be harvested. In contrast, many overseas pasture systems export large quantities of pasture from paddocks to other farms or to be fed indoors during winter. For example, the average annual C export in harvested pasture from six different pasture systems in Europe was 3,072 kg C  $\text{ha}^{-1}$  (Soussana *et al.*, 2007).

On dairy farms, transfer of C in the form of dung and urine via animals from paddocks to raceways and the milking shed is another pathway of C loss from the paddocks (and thus the soil) but not the whole farm system. Usually effluent from the dairy shed is returned to a proportion of the farm via effluent irrigation.

### 2.5.2.3 Methane

Methane is a major contributor to greenhouse gas emissions from New Zealand, accounting for 38% of total emissions compared to only 5 – 10% for most other industrialized countries (Waghorn and Woodward, 2006). The reason for this is that New Zealand has a large population of grazing ruminant animals compared to many other countries, with 88% of total methane emissions coming from animals of which 98% is due to digestion processes within the animal (Waghorn and Woodward, 2006).

Although methane emissions from farms are relatively large in terms of global warming potential, losses of C in methane from pasture systems are relatively small in comparison to losses of C via respiration. Laubach and Kelliher (2004) measured methane emissions from a herd of grazing dairy cows in spring, mid-summer, and late summer and found that emissions averaged  $0.247 \text{ kg CH}_4\text{-C cow}^{-1}\text{day}^{-1}$ . At three similar times of the year, Robertson and Waghorn (2002) measured similar values which averaged  $0.259 \text{ kg CH}_4\text{-C cow}^{-1}\text{day}^{-1}$ . A study by Woodward *et al.* (2004) found that lactating dairy cows grazing ryegrass pasture produced  $0.296 \text{ kg CH}_4\text{-C cow}^{-1}\text{day}^{-1}$ . The average of these three studies was  $0.258 \text{ kg CH}_4\text{-C cow}^{-1}\text{day}^{-1}$  which is equivalent to  $267 \text{ kg CH}_4\text{-C ha}^{-1} \text{ y}^{-1}$  (based on national the dairy farm stocking rate of  $2.81 \text{ cows ha}^{-1}$  during the 2006/2007 season (Livestock Improvement, 2007)).

New Zealand is a world leader in measuring methane emissions from grazing ruminants and the Ministry for the Environment (2007) has compiled a large database of values (many of these studies are unpublished). Results from many of the studies have been reported in terms of grams of methane produced per kilogram of dry matter ingested with average values of 16.2 and 15.6 g  $\text{CH}_4\text{-C}$  produced per kg DM intake for dairy cattle and adult sheep respectively (Ministry for the Environment, 2007). Using these values combined with average

production data (milk, meat, wool), the Ministry for the Environment (2007) calculated that on average dairy cows emitted  $59 \text{ kg CH}_4\text{-C cow}^{-1} \text{ y}^{-1}$ , beef cattle  $43 \text{ kg CH}_4\text{-C cow}^{-1} \text{ y}^{-1}$  and sheep  $8.2 \text{ kg CH}_4\text{-C sheep}^{-1} \text{ y}^{-1}$ . Based on the national average dairy farm stocking rate of  $2.81 \text{ cows ha}^{-1}$  (Livestock Improvement, 2007) average C loss via methane emission would be about  $166 \text{ kg C ha}^{-1}$  on dairy farms. Stocking rates on sheep and beef farms vary considerably, but an average medium hill country farm in the Waikato would have a stocking rate of  $\sim 12$  stock units per  $\text{ha}^{-1}$  (equivalent to  $12 \text{ sheep ha}^{-1}$ ) and therefore losses of C via methane would be about  $100 \text{ kg C ha}^{-1} \text{ y}^{-1}$ .

Methane production from dairy cows predicted using the Overseer® model (a model developed by AgResearch, that calculates nutrient and greenhouse gas budgets for New Zealand farms) was  $199 \text{ kg CH}_4\text{-C ha}^{-1} \text{ y}^{-1}$ , when using the average New Zealand dairy farm stocking rate of  $2.81 \text{ cows ha}^{-1}$  and average milk solid production of  $934 \text{ kg ha}^{-1} \text{ y}^{-1}$  (Livestock Improvement, 2007).

There was a discrepancy between methane production values calculated using data from different sources. Direct measurements of  $\text{CH}_4\text{-C}$  production from three studies averaged  $267 \text{ kg CH}_4\text{-C ha}^{-1} \text{ y}^{-1}$ , compared to  $166 \text{ kg CH}_4\text{-C ha}^{-1}$  using data from the Ministry for the Environment (2007) and  $199 \text{ kg CH}_4\text{-C ha}^{-1} \text{ y}^{-1}$  when using the Overseer® model (all values calculated based on stocking rate of  $2.81 \text{ cows ha}^{-1}$ ). Higher values reported from direct measurements than the two calculated values (MFE and Overseer®) would have probably been caused because the direct measurements were only made when cows were lactating and pasture intakes would have been higher than during the winter when cows were dry. It is likely that lower methane emissions when cows are dry has been factored into the Ministry for the Environment (2007) and Overseer® model calculations, but this has not been explicitly stated. Although these discrepancies are important for greenhouse gas reporting, they are less important for C budgeting because the differences between the values are relatively small in comparison to other C flows (e.g. photosynthesis and respiration). A published review of methane measurements (from published and unpublished studies) would assist researchers who are trying to construct C and greenhouse gas budgets for New Zealand pastoral systems.

#### 2.5.2.4 Leaching

Globally, there is little information on leaching losses of C from grazed pasture (McTiernan *et al.*, 2001). Hope *et al.* (1994) completed a review of C export in river waters (from catchments with a range of vegetation types) and found that DOC was the main form of C and that most catchments exported between 10 and 100 kg C ha<sup>-1</sup> y<sup>-1</sup>. Only seven temperate grasslands were represented in the review with an average DOC export of approximately 5.8 kg C ha<sup>-1</sup> y<sup>-1</sup>. In an ungrazed grassland in Germany, Don and Schulze (2008) measured DOC leaching of 8 kg C ha<sup>-1</sup> y<sup>-1</sup> in a clay rich Vertisol and 55 kg C ha<sup>-1</sup> y<sup>-1</sup> in an Arenosol with low clay content. The lower leaching from the Vertisol was also accompanied by a time lag of several months, which was suggested to be due to greater adsorption of DOC to clay minerals which lead to longer retention and greater mineralization. McTiernan *et al.* (2001) made measurements of dissolved organic carbon (DOC) leaching from ten, 1 ha hydrologically isolated lysimeters (that were grazed by steers) over a period of two months. They found that total C export varied from 42 to 118 kg C ha<sup>-1</sup> and was significantly higher from plots that had not been tile drained. On the poorly drained plots, DOC leaching losses were significantly higher from lysimeters that had received N fertilizer.

There is also very little information on C leaching in New Zealand. Sparling *et al.* (2006) conducted an outdoor barrel lysimeter study and found that averaged over four years a pumice soil leached 39.8 kg C ha<sup>-1</sup> y<sup>-1</sup>, a gley soil 56.3 kg C ha<sup>-1</sup> y<sup>-1</sup>, a recent soil 241.8 kg C ha<sup>-1</sup> y<sup>-1</sup> and an allophanic soil 36.5 kg C ha<sup>-1</sup> y<sup>-1</sup>. Ghani *et al.* (2008) carried out a lysimeter study on six different New Zealand soils collected from under intensive dairying. The soils were placed in a growth chamber and incubated at 20 °C, with 14 hours of light and 10 hours of darkness and leached about every 3 – 5 weeks during a six month period. Extrapolated to an annual basis DOC leaching was 1,834 and 1,217 kg C ha<sup>-1</sup> y<sup>-1</sup> from two gley soils and between 332 and 861 kg C ha<sup>-1</sup> y<sup>-1</sup> from four allophanic soils. These results must be treated with caution because the experiment was not carried out under field conditions or for a complete year. However, the study of Ghani *et al.* (2008) does demonstrate that DOC leaching could be a significant output of C, particularly from poorly drained soils, which is consistent with Sparling *et al.* (2006) and McTiernan *et al.* (2001).

These studies demonstrate that C leaching losses can vary considerably, with soil type and/or drainage having a significant impact on the amount of C leached. Lysimeter studies may not tell the whole story of C losses via leaching because it is not known how much of the C leached below lysimeter collection points is lost via lateral flow, how much is stored in lower soil (or rock) horizons and how much is respired in the vadose zone and groundwater and then lost to the atmosphere. Values of C export in river water calculated by Hope *et al.* (1994) were generally lower than those measured from leaching studies, which suggests storage and/or respiratory losses of C below lysimeter collection points may occur.

#### **2.5.2.5 Erosion**

C losses via erosion can vary significantly depending on topography and soil type. In highly erodible soft rock hill country in the Hawkes Bay, Page *et al.* (2004) calculated that  $500 \text{ kg C ha}^{-1} \text{ y}^{-1}$  entered a nearby lake from landslide erosion and  $440 \text{ kg ha}^{-1} \text{ y}^{-1}$  from sheet wash erosion (since the establishment of pastoral farming 114 years previous). Page *et al.* (2004) explained that these losses would be the upper limit of C losses via erosion, but that net C losses would have been much lower because  $\sim 600 \text{ kg C ha}^{-1} \text{ y}^{-1}$  was sequestered on old slip scars. Hunt *et al.* (2004) calculated that wind induced topsoil erosion caused a C loss of  $20 \text{ kg C ha}^{-1} \text{ y}^{-1}$  in a sparse (43% bare ground) seasonally dry tussock grassland. Based on monthly measurements of suspended sediment loads in 44 rivers throughout New Zealand, Scott *et al.* (2006) modelled C export in sediment for all of New Zealand. Excluding the soft rock country described above, they predicted that losses of C from pasture land ranged from  $0 - 50 \text{ kg ha}^{-1} \text{ y}^{-1}$  with the majority of the flatter pastoral areas having C losses of  $<20 \text{ kg ha}^{-1} \text{ y}^{-1}$ . These results suggest that except for erosion prone hill country, C losses via erosion are small compared to other C loss pathways.

## **2.6 Net soil carbon balance under grazed pasture**

This section will review studies that have reported on changes in soil C in pastoral systems. The first part will present results from studies that have directly measured changes in soil C and will focus on long term New Zealand pasture

systems. The second part will present results from eddy covariance (EC) studies and due to the paucity of such studies in New Zealand will also look to international studies.

### **2.6.1 *Measured changes in soil C content***

Two of the earliest studies investigating changes of C in New Zealand pastoral soils were conducted by Walker *et al.* (1959) and Jackman (1964). Both studies used the chronosequence approach to investigate how C (and other nutrients) changed with time after new high producing pastures were sown following ploughing of old poor producing pastures or scrub. In both studies, there was an increase in percent C in the surface soil (0 – 75 mm) over time. Jackman (1964) found that the increase in C was not linear, being rapid at first and then declining with time as a new steady state was reached between 7 – 40 years after ploughing. In both studies, there were no consistent changes in percent C below 75 mm. Walker *et al.* (1959) calculated that the average annual increase after 25 years for the 0 – 200 mm depth was 125 kg ha<sup>-1</sup> but noted that 25 years of dairy cattle grazing had caused significant compaction which would have likely resulted in an overestimate of C accumulation. Jackman (1964) did not present changes in C on a mass basis although bulk density was reported. Schipper and Sparling (submitted-a) reworked the Jackman (1964) data and found that all individual soils showed a significant increase in C content for the 0 – 75 mm depth, but there was only a significant increase in C in one soil when the C content was summed to 300 mm depth. This effect was caused because changes in soil C at the surface were masked by non-changes at depth. Averaged over all 10 soils (for the 0 – 75 mm depth) there was an increase of 1,070 kg C ha<sup>-1</sup> during the first five years after pasture establishment, which declined to 270 kg C ha<sup>-1</sup> between 5 – 25 years after establishment and then 90 kg C ha<sup>-1</sup> between 25 – 50 years after establishment (Schipper and Sparling, submitted-a). The rate of increase was almost exactly the same for the 0 – 300 mm depth which indicates that almost all of the change in C content occurred in the 0 – 75 mm depth of soil.

The effect of fertilization on soil C has been investigated in a number of studies. McIntosh *et al.* (1999) found that fertilization (with sulphur-superphosphate) and oversowing (with clover and grass seed) native tussock grassland (grazed by



sheep) in the high country of the South Island increased soil C (0 – 150 mm depth) by 7,000 kg C ha<sup>-1</sup> over a 19 year period (368 kg C ha<sup>-1</sup> y<sup>-1</sup>). The increase was associated with a decrease in the amount of bare ground and increased pasture productivity. Investigation of changes in C at long term phosphate fertilizer trials on hill country pastures in the southern Hawke's Bay (Lambert *et al.*, 2000) and in the Waikato (Schipper and Sparling, submitted-b) revealed that phosphate fertilizer application rate had no effect on soil C content in the surface soil (0 – 75 mm). However, the study of Lambert *et al.* (2000) did find that soil C content declined by about 200 kg ha<sup>-1</sup> y<sup>-1</sup> during the study (16 years) in both the high and low fertilizer application treatments. At the Whatawhata research station in the Waikato, Schipper and Sparling (submitted-b) found that C increased significantly on easy (1,560 kg ha<sup>-1</sup> y<sup>-1</sup>) and steep slopes (1,060 kg ha<sup>-1</sup> y<sup>-1</sup>) during the first 6 years of the trial. During the last 17 years of the trial there was a non-significant decline in C content on easy slopes and a significant decline of 450 kg ha<sup>-1</sup> y<sup>-1</sup> on the steep slopes. Schipper and Sparling (submitted-b) attributed the decline in C in the latter part of the study to a large decrease in pasture production caused by a series of dry summers. At a N fertilizer rate experiment in the Waikato, Ghani *et al.* (2003) found that soil C content was 6.3%, 5.3% and 4.7% after 5 years of N application rates of 0, 200 and 400 kg N ha<sup>-1</sup> y<sup>-1</sup> respectively. The standard error of the differences of means was 0.38% indicating these were significant differences. However, there was only one sampling and it was unclear whether soil C levels were initially the same in all treatments. The combination of irrigation and establishment of new high producing pastures on a previously dryland pasture on the Canterbury plains caused an initial increase in soil C which then levelled off after about 15 – 16 years (Nguyen and Goh, 1990). In the same trial, soils fertilized with superphosphate attained a significantly higher steady state C content than unfertilized soils (Nguyen and Goh, 1990).

In 1992, Tate *et al.* (1997) re-sampled surface soils at 43 sites under pastoral agriculture that had been previously sampled 30 – 50 years earlier. Tate *et al.* (1997) only measured percent C and found that two sites with peaty soils showed large decreases in C% which was attributed to drainage after to the initial sampling. There was no systematic temporal change in C% at the other sites. Tate (2005) developed an IPCC-based carbon monitoring system for New Zealand and

assumed (based on the limited data available) that C in NZ pastoral soils was at or near steady state. More recently, Schipper *et al.* (2007) re-sampled 31 flat or gently rolling pastoral sites (mainly dairy farms) throughout New Zealand that had been sampled 17 – 30 years previously (sites were sampled to about 1 m depth). They found that on average these soils had lost  $1,060 \text{ kg C ha}^{-1} \text{ y}^{-1}$ . These losses were of similar magnitude to those reported by Bellamy *et al.* (2005) for soils under pasture in England and Wales, during a similar time period.

These studies do not reveal any consistent trend of changing soil C with time under New Zealand pastoral agriculture. Some studies report increases, some decreases and some no change. In studies that report changes, the cause of the changes are often poorly understood. One of the reasons for this is that to detect changes in soil C, long time periods are required and changes in land management are often poorly documented. The length of time required to detect changes in soil C means direct sampling has limited value if results are required in the short term. New approaches are required that will be able to detect smaller differences over shorter time scales and determine the effect of specific management practices on soil C. The eddy covariance approach discussed in the next section is one possible tool that could be used to determine the factors driving changes in soil C.

### **2.6.2 Eddy covariance studies**

While the EC approach has been widely used around the world the majority of studies have been on forest ecosystems with much less emphasis on grasslands (Falge *et al.*, 2002; Soussana *et al.*, 2007), particularly intensively grazed grasslands such as those common in New Zealand. The previous section detailed how in some grasslands in New Zealand soil C was increasing, in others soil C remained constant and in others there were large losses of C. Results from global EC studies show a much clearer trend with the majority of studies showing that grasslands are a net  $\text{CO}_2$ -C sink. For example, Gilmanov *et al.* (2007) presented results from 20 European grasslands, 16 of which were  $\text{CO}_2$ -C sinks, with the average annual NEE of all 20 sites being  $-1503 \text{ kg C ha}^{-1} \text{ y}^{-1}$ . Of the four sites that were net sources of  $\text{CO}_2$ -C, two had peaty soils and the other two sites experienced drought stress during the years of measurements (Gilmanov *et al.*, 2007). Soussana *et al.* (2007) presented results from eight of the sites reported by

Gilmanov *et al.* (2007) but also calculated the net ecosystem carbon balance (NECB) (which included outputs of C in methane and harvested biomass and inputs of C in manure). Even with these other C pathways included these eight grasslands were still net sinks of C (average  $-1040 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ) (although they did not include DOC leaching or outputs in animal products). Table 2.1 presents a summary of the carbon balance from a number of EC studies of managed grasslands and in general results are consistent with the findings of Gilmanov *et al.* (2007) that grasslands are net sinks of C, even when including exports in product (milk, meat, plant biomass), methane and leaching losses.

Under specific circumstances, however, grasslands can become net sources of C. As expected, grasslands on drained peat soils lose carbon (Nieveen *et al.*, 2005; Gilmanov *et al.*, 2007; Veenendaal *et al.*, 2007) because increased oxygen inputs increase peat decomposition. Sites which export large amounts of silage or hay can also be net C sources (Skinner, 2008). Severe drought conditions cause grasslands to become a net source of C, or at least decrease the amount of C sequestered (Hunt *et al.*, 2004; Ammann *et al.*, 2007; Aires *et al.*, 2008) (Table 2.1). Hunt *et al.* (2004) and Xu and Baldocchi (2004) found that the timing of precipitation was more important than total precipitation in the relatively dry environments in which they were working. For example, Xu and Baldocchi (2004) studied an annual grassland in California and during the second year of their study, winter rain was delayed causing the annual grass to germinate a month later than usual which shortened the growing season and thus annual GPP. In addition, a large storm after the grass had senesced meant that soil moisture was higher during summer which increased respiration compared to the first year (Xu and Baldocchi, 2004). It has also been clearly demonstrated that even small rainfall events after dry periods can cause large (though short-term) increases in respiration (Hunt *et al.*, 2004; Xu and Baldocchi, 2004; Aires *et al.*, 2008; Jenerette *et al.*, 2008). Small rain events cause relatively large net losses of C because they provide sufficient water to stimulate microbial activity in the surface soil but not enough water to penetrate to the root zone and stimulate photosynthesis (Xu and Baldocchi, 2004; Baldocchi, 2008). Harvesting pasture or grazing has also been shown to cause grasslands to flip from being a C sink to a C source almost instantly due to removal of photosynthetic material, although this

effect is usually only short lived (Novick *et al.*, 2004; Nieveen *et al.*, 2005; Jaksic *et al.*, 2006; Veenendaal *et al.*, 2007).

Table 2.1. Summary of eddy covariance studies of managed grasslands. Values are in kg C ha<sup>-1</sup> y<sup>-1</sup>. Negative values represent inputs and positive values outputs from the system.

Reference	Location	Vegetation	Management	Year(s)	Mean annual temp (°C)	Rain (mm y <sup>-1</sup> )	GPP <sup>a</sup>	TER <sup>a</sup>	NEE <sup>a</sup>	Supplement & manure inputs	Supplement outputs	Animal product outputs	Methane from animals	Other	NECB <sup>a</sup>
Jaksic <i>et al.</i> (2006)	Ireland	Perennial ryegrass	Dairy pasture, grazed and cut for silage	2002		1,785	-16,730	14,800	-1,930	<sup>b</sup>		440	165	1,089 <sup>c</sup>	-240
				2003		1,185	-17,180	14,600	-2,580	<sup>b</sup>		440	165	1,089 <sup>c</sup>	-890
Byrne <i>et al.</i> , (2007)	Ireland	Perennial ryegrass	Dairy pasture, grazed and cut for silage	2004		1,470*			-2,900	-540		280 <sup>d</sup>	115	945 <sup>e</sup>	-2,100
Nieveen <i>et al.</i> (2005) <sup>f</sup>	New Zealand	Perennial ryegrass and white clover	Rotationally grazed dairy pasture	2002		1,281	13,486	13,531	45			738	278		1061
Veenendaal <i>et al.</i> (2007) <sup>f</sup>	Netherlands	Perennial ryegrass	Cut three times, grazed 2-3 times	2005	9.8*	793*			1,339	-1540	4100		350		4230
Hunt <i>et al.</i> (2004)	New Zealand	Tussock, <i>Heiracium</i> (43% bare ground)	Extensive grazing by sheep, unfertilized	1998	9.9	446			90						
				1999	9.2	933			-410						
Allard <i>et al.</i> (2007)	France	Permanent pasture	Continuous grazing (May - Oct) with heifers, stocking rate to maintain sward at 60 mm	2002		1,128	-17,240	16,740	-500			16	94		-390
				2003		1,177	-14,980	14,070	-910			16	90		-800
				2004		807	-15,600	14,050	-1,550			20	109		-1,420
		Permanent pasture	As above except half stocking rate	2002		1,128	-16,940	15,820	-1,120			7	51		-1,060
				2003		1,177	-14,410	13,920	-490			8	46		-440
				2004		807	-14,080	13,450	-640			13	54		-570

<sup>a</sup> GPP, gross primary production; TER, total ecosystem respiration; NEE, net ecosystem exchange; NECB, net ecosystem carbon balance.

<sup>b</sup> Concentrate feed was imported but they did not include it.

<sup>c</sup> Includes 1039 kg of respiration from cows in dairy shed and when housed during winter and 50 kg of DOC leaching.

<sup>d</sup> They did not include C in lactose in their calculation.

<sup>e</sup> Includes 775 kg of respiration from cows in dairy shed during milking and when housed during winter. The remainder CH<sub>4</sub>-C from manure spreading and DOC leaching.

<sup>f</sup> Drained peat soil.

\*Mean annual average (not specific to year of study).

Table 2.1 – continued

Reference	Location	Vegetation	Management	Year(s)	Mean annual temp (°C)	Rain (mm y <sup>-1</sup> )	GPP	TER	NEE	Supplement & manure inputs	Supplement outputs	Animal product outputs	Methane from animals	Other	NECB
Ammann <i>et al.</i> (2007)	Switzerland	Permanent pasture (7 Pasture sp.)	Intensive (cut for silage 4 times 200 kg N ha <sup>-1</sup> y <sup>-1</sup> )	2002	9.6	1,479	-21,590	14,900	-6,690	-590	4,620				-2,660
				2003	9.6	895	-17,730	15,580	-2,150	-590	2,410				-330
				2004	8.9	1,158	-20,560	15,390	-5,170	-220	4,010				-1,380
		Permanent pasture (30 Pasture sp.)	Extensive (cut for silage 3 times)	2002	9.6	1,479	-17,140	13,620	-3,520		3,800				280
				2003	9.6	895	-17,500	16,780	-710		2,190				1,480
				2004	8.9	1,158	-20,750	17,360	-3,390		3,350				-40
Skinner (2008)	Pennsylvania	Grass based pasture	Rotationally grazed by cattle and harvested for silage	2003-2006	9.7*	1,014*			270	-400	1,260	Not included			1130
		Alfalfa based pasture	Rotational grazing by cattle and harvested for silage	2003-2006	9.7*	1,014*			-650	-130	1,260	Not included			480
Aires <i>et al.</i> (2008)	Portugal	C3/C4 grassland	Grazed by sheep (Oct-Feb)	2004		364	-5,240	5,730	490						
Novick <i>et al.</i> (2004)	North Carolina	Predominantly C4 grass pasture	Mowed annually in summer for hay	2001	15.5	1,145	-12,020	12,990	970						
Xu and Baldocchi (2004)	California	Annual grassland, rocky silt loam (Lithic xerorthents)	Grazed but not specified by what	2001	16.3*	567	-867	735	-1,320						
				2002	16.3*	494	-729	758	290						

\*Mean annual average (not specific to year of study)

Table 2.1 – continued.

Reference	Location	Vegetation	Management	Year(s)	Mean annual temp (°C)	Rain (mm y <sup>-1</sup> )	GPP	TER	NEE	Supplement & manure inputs	Supplement outputs	Animal product outputs	Methane from animals	Other	NECB
Soussana et al., (2007)	Hungary	Semi-natural grassland	Grazing	1	10.5*	500*			-130				8		-120
				2	10.5*	500*			-1,250				15		-1,240
	Scotland	Intensive permanent grassland	Grazing and cutting	1	8.8*	638*			-3,840	-30	2,200		59		-1,610
				2	8.8*	638*			-3,020	-30			49		-3,000
	Ireland	Sown grass/clover	Grazing and cutting	1	9.4*	824*			-3,720		2,710		51		-960
				2	9.4*	824*			-2,140		4,760		39		2,660
	France	Semi-natural grassland (intensive)	Grazing	1	8*	1,313*			-500				91		-410
				2	8*	1,313*			-1,120				90		-1,030
	France	Semi-natural grassland (extensive)	Grazing	1	8*	1,313*			-910				54		-860
				2	8*	1,313*			-490				46		-440
	Netherlands	Intensive permanent grassland	Grazing and cutting	1	10*	780*			n.d	-1,040	2,370		104		n.d
				2	10*	780*			-1,770	-800	2,200		42		-330
	Italy	Semi-natural grassland	Grazing	1	6.3*	1,200*			-4,640				21		-4,620
				2	6.3*	1,200*			-2,550				21		-2,530
	Switzerland	Sown grass/clover (intensive)	Cutting	1	9*	1,109*			-4,190	-1,060	4,600				-650
				2	9*	1,109*			-4,140	-290	2,400				-2,030
	Switzerland	Sown grass/clover (extensive)	Cutting	1	9*	1,109*			-3,520		3,800				280
				2	9*	1,109*			-2,930		2,100				-830
	Denmark	Barley - grass rotation	Cutting	1	9.2*	731*			-310	-14,500	4,060				-10,750
				2	9.2*	731*			-3,730	-13,650	2,590				-14,790

\*Mean annual average (not specific to year of study)

## **2.7 The effect of physical disturbance on soil C and CO<sub>2</sub> emissions**

It is well documented that physical disturbance of soil can cause losses of soil carbon (Tisdall and Oades, 1982; Grandy and Robertson, 2007). Two types of physical disturbance that occur on New Zealand pastoral soils are cultivation and pugging (the plastic deformation of wet soil by animal hooves, Greenwood and McKenzie, 2001). These two types of physical disturbance will be the focus of the next section. Long term cultivation has been shown to cause large losses of soil C (Davidson and Ackerman, 1993), but there is much less information on the effect that occasional cultivation (such as occurs in pastoral systems) has on soil C (Conant *et al.*, 2007). Even less is known about the effect that pugging damage by grazing animals has on soil C and CO<sub>2</sub> emissions.

### **2.7.1 Cultivation**

#### **2.7.1.1 Effect on soil C**

Long term cultivation generally causes a 20 – 60% reduction in soil C content compared to original C contents under permanent vegetation (Davidson and Ackerman, 1993; Murty *et al.*, 2002; Lal, 2004). Losses are greatest in the first few years after initial cultivation and then decline as the soil moves to a new equilibrium (Davidson and Ackerman, 1993; Haynes and Tregurtha, 1999). C loss following cultivation is generally attributed to the physical disturbance of cultivation damaging soil aggregates which exposes previously protected C to microbial decomposition (Tisdall and Oades, 1982; Six *et al.*, 2004; Grandy and Robertson, 2007). The actual cause of C loss in cultivated systems has recently become controversial. Baker *et al.* (2007) proposed that a reduction in C input to the soil under cropping regimes due to fallow periods (which are not present under perennial vegetation) may contribute to loss of soil C from cropping systems. Lower C allocation to roots in crop plants (Kuzyakov *et al.*, 2001) and large C export in product (Anthoni *et al.*, 2004) are other possible causes of C decline in cropping soils.



Cultivation of pastoral soils is infrequent, generally only being used to establish forage crops and sow new pasture (e.g. about every 10 – 15 years on New Zealand dairy farms). It might be expected that such infrequent cultivation would not have much effect on soil C content. However, it has been clearly demonstrated that the largest losses of soil C occur in the first few years after initial cultivation (Davidson and Ackerman, 1993; Haynes and Tregurtha, 1999) and therefore periodic cultivation could contribute to a loss of soil C if there was not sufficient time between cultivation events for C levels to be replenished. This was illustrated by the study of Martel and Mackenzie (1980) on dairy farm systems in Canada. Traditional farm systems were based on a 5 year rotation, with one year under a cereal crop followed by 4 years under pasture (grazed or cut for hay). Under this management regime soil C levels (A and B horizons) were about 33% lower than under virgin forest. If the period of pasture was extended to 10 years, soil C levels rose to the same level found in the virgin forest. However, if the soils were continuously row cropped, C levels declined by up to 60% relative to the virgin forest soil. This study clearly illustrates the need to understand the effect of occasional cultivation on soil C in pastoral systems.

Conant *et al.* (2007) reviewed the effect that periodic tillage had on soil C stocks, which revealed that there was very limited data on this topic and definitive conclusions could not be made. An indication of the approximate magnitude of C loss following a single cultivation event in New Zealand pastoral systems can be estimated by calculating the rate of C loss after a few years of cultivation when changes in soil C content are large enough to detect (Table 2.2). However, it must be noted that losses will likely be greatest after the first cultivation event and then decline with time and therefore such an approach will likely underestimate the loss of C loss after the initial cultivation event. Table 2.2 shows that the rate of C loss ranges between about 100 and 4,000 kg C ha<sup>-1</sup> y<sup>-1</sup> after 2 – 7 years of cultivation, depending on the cropping system and soil type.

**Table 2.2:** Total C loss and rate of C loss, following conversion of permanent pasture (via cultivation) to various cropping systems in New Zealand.

Reference	Soil	Sampling depth (mm)	Treatment	Treatment duration (years)	Original TC (kg/ha/y)	New TC (kg/ha/y)	Total change in C (kg/ha/y)	Rate of C change (kg /ha/y)
Shepherd <i>et al.</i> (2001)	Manawatu silt loam	0-200	Maize or barley cropping	4	85,200	75,600	−9,600	−2,400
	Kairanga silty clay loam	0-200	Maize or barley cropping	4	90,800	77,500	−13,300	−3,325
	Moutoa humic clay	0-200	Maize or barley cropping	4	164,300	163,800	−0,500	−125
	Patumahoe clay	0-200	Double cropped (Barley and brassica)	40	92,500	46,300	−46,200	−1,155
	Egmont silt loam	0-200	Double cropped with vegetables	20	135,800	131,800	−4,000	−200
Crush <i>et al.</i> (1992)	Kairanga silt loam	0-150	Permanent pasture	3	52,100	53,500	1,400	467
		0-150	Cultivated cereal	3	50,100	40,470	−9,630	−3,210
Saggar <i>et al.</i> (2001b) <sup>a</sup>	Marton silt loam	0-100	7 years cropping	7	44,900	28,300	−16,600	−2,371
		0-100	20 year cropping	20	44,900	27,200	−17,700	−885
	Kairanga silty clay loam	0-100	5 years maize cropping	5	51,400	37,100	−14,300	−2,860
		0-100	16 years maize cropping	16	51,400	37,300	−14,100	−881
		0-100	34 years maize cropping	34	51,400	24,600	−26,800	−788
Aslam <i>et al.</i> (2000) <sup>b</sup>	Ohakea silt loam	0-100	2 years cropping (mouldboard ploughing to 200 mm)	2	35,3	27,600	−7,700	−3,850
		0-100	2 years cropping (no till)	2	35,3	34,000	−1,300	−650

<sup>a</sup> Calculated from bulk density and C concentrations as mass values were not reported. Bulk density increased with length of cropping duration. Assuming C content in pasture treatment had not changed with time.

<sup>b</sup> Assuming C content in pasture treatment had not changed with time.

### 2.7.1.2 Effect on soil CO<sub>2</sub> emissions

Measuring CO<sub>2</sub> emissions immediately after cultivation is one way of quantifying C loss caused directly by cultivation. A major advantage of measuring CO<sub>2</sub> emissions compared to measuring changes soil C is that smaller C losses can be detected. The temporal pattern of C losses following cultivation can also be determined and environmental or management factors that influence C loss can be identified. Chamber systems are the most commonly used method to measure the effect that cultivation has on CO<sub>2</sub> emissions, although measurements at the paddock scale using the eddy covariance technique have also been made (Anthoni *et al.*, 2004).

Large peaks (up to 80 kg ha<sup>-1</sup> h<sup>-1</sup>) in CO<sub>2</sub> emissions have been measured seconds to minutes after cultivation (Reicosky and Lindstrom, 1993; Reicosky *et al.*, 1997; Ellert and Janzen, 1999; Alvaro-Fuentes *et al.*, 2007). However, these large flushes of CO<sub>2</sub> are usually short lived. For example, two hours after cultivation Reicosky *et al.* (1997) found that fluxes had decreased to 20% of the flux immediately after cultivation and after 24 hours fluxes had declined to almost the level of the un-cultivated control plots. These large flushes of CO<sub>2</sub> are thought to be primarily caused by the physical release of air, rich in CO<sub>2</sub>, from soil pores rather than an increase in microbial activity (Reicosky *et al.*, 1997; Ellert and Janzen, 1999). This hypothesis was supported by Wuest *et al.* (2003) who found that physical disturbance caused an increase in CO<sub>2</sub> emissions from a previously sterilized soil with the increase in emissions being of similar magnitude to that following disturbance of un-sterilized soil. These short term, largely abiotic flushes of CO<sub>2</sub> following cultivation are probably of little concern in terms of soil C content, because the CO<sub>2</sub> lost from the soil will be replaced as a new equilibrium CO<sub>2</sub> concentration in the soil is reached. Only if the loss of CO<sub>2</sub> from the soil was caused by an increase in microbial respiration would this cause a long-term net loss of C from the soil.

Longer term studies of CO<sub>2</sub> emissions following cultivation report a range of results, some studies showing that cultivation increased CO<sub>2</sub> emissions compared to un-cultivated soil, while others show decreased CO<sub>2</sub> emissions (Table 2.3). Aslam *et al.* (2000) measured CO<sub>2</sub> emissions using chambers for almost a year (about once per month) from permanent pasture in New Zealand (grazed by

sheep) and paddocks double-cropped with maize and oats following mouldboard ploughing or using no tillage (paddocks previously permanent pasture). CO<sub>2</sub> emissions were always higher from the pasture than cultivated paddocks at each measuring time throughout the year. Average CO<sub>2</sub>-C emissions over the study were 63, 60 and 92 kg C ha<sup>-1</sup> day<sup>-1</sup> from the conventionally tilled, non-tilled and pasture treatments respectively (Table 2.3). During a short 21 day study, Yamulki and Jarvis (2002) found that average CO<sub>2</sub> emissions were 76 kg C ha<sup>-1</sup> day<sup>-1</sup> from a pasture mowed 1 week prior to measurements compared to 43 kg C ha<sup>-1</sup> day<sup>-1</sup> from the same pasture that had been mowed, sprayed with herbicide and cultivated to 200 mm depth. Higher CO<sub>2</sub> emissions from the pasture treatments in these two studies was likely caused by a greater contribution of CO<sub>2</sub> from root respiration than in the cultivated paddocks (Craine *et al.*, 1999; Wan and Luo, 2003; Byrne and Kiely, 2006).

In contrast, Grandy and Robertson (2006) found that CO<sub>2</sub>-C emissions from a never tilled field averaged 32.2 kg ha<sup>-1</sup> day<sup>-1</sup> compared to 48.6 kg ha<sup>-1</sup> day<sup>-1</sup> from the same field cultivated by mouldboard ploughing. They attributed the higher emissions from the cultivated plots to higher soil temperatures and the destruction of aggregates which increased the amount of labile C available for microbial decomposition.

Most studies of CO<sub>2</sub> emissions in relation to cultivation have been comparisons between tillage treatments. In general, cultivation by mouldboard ploughing increases CO<sub>2</sub> emissions compared to no-till (Reicosky and Lindstrom, 1993; Bauer *et al.*, 2006; Chatskikh *et al.*, 2008), although this is not always the case. For example Quincke *et al.* (2007) found that cultivation of a previously no-till soil in Nebraska using mouldboard ploughing to 200 mm depth did not increase CO<sub>2</sub> emissions compared with no-till, but chisel ploughing to 200 mm did increase CO<sub>2</sub> emissions. This difference was attributed to the fact that mouldboard ploughing inverted the soil, burying most of the labile carbon at depth where temperature was consistently lower, which would have limited microbial activity. A similar effect was also noted by Aslam *et al.* (2000) after the autumn tillage in their study (described above).

While it is important to understand the rate of CO<sub>2</sub>-C loss following cultivation of pastoral soils, this is only half the story with regard to the long term soil C balance. The rate of C accumulation and new steady state following establishment of new permanent pastures is also important. For the first few months after new pastures have germinated, growth is usually lower than established pastures and therefore C inputs to the soil are likely to be lower. However, once new pastures have become fully established they are usually more productive than old pastures and therefore C inputs to the soil may be greater than prior to cultivation. Measurement of CO<sub>2</sub> emissions using chamber techniques cannot quantify the total C loss due to cultivation because photosynthetic inputs of C to the system are not measured. Using the EC technique, or directly measuring changes in soil C over time may be better approaches if the long term effect of cultivation on soil C is the aim of the study. However, in dairy farm systems, cultivated areas will often be too small to meet fetch requirements of EC systems (see Section 2.4.2.2) and direct measurement of soil C also has many limitations (see Section 2.4.1).

Spraying of existing pastures followed by direct drilling, is another common way in which pastures and crops are sown. Pasture renewal often occurs in conjunction with a summer crop such as turnips or maize in which case two cultivation (or direct drilling) events take place. The rate of pasture renewal will Quantifying soil C loss due to these different practices would improve our understanding of the C balance of dairy farm systems.

**Table 2.3:** Average respiration rates and total C respired from cultivated and control treatments, collated from a number of studies around the world.

Reference	Location	Method	Soil	Management	Year/ season	No. of measurements	Study duration (days)	Average respiration (kg ha <sup>-1</sup> day <sup>-1</sup> )	Total C respired (kg ha <sup>-1</sup> )
Aslam <i>et al.</i> (2000)	New Zealand	Alkali trap	Typic andoaqualf	Permanent pasture grazed by sheep		12	341	92	31372
				Mouldboard ploughing to 200 mm to prior to maize sowing in spring and oats in winter		12	341	63	21483
				No till (same crops as above)		12	341	60	20460
Yamulki & Jarvis (2002)	Devon (UK)	Closed chamber	Clay loam (poorly drained)	Permanent pasture cut one week prior to experiment		6	21	76.2	1600
				Above pasture sprayed and cultivated to 200 mm with a blade rotovator		6	21	42.9	900
Grandy and Robertson (2006)	Michigan (USA)	Static chamber	Mesic Typic Hapludalfs	Never tilled field	2002	13	60	36.7	2448
					2003	10	138	27.6	4074
					2004	19	198	32.2	6653
				Tilled once with a mouldboard plough then disc harrowed and left fallow (let weeds grow)	2002	13	60	56.4	3269
					2003	10	138	48.48	6690
					2004	19	198	41.04	8554
Quinke <i>et al.</i> (2007)	Nebraska (USA)	Closed chamber	Mesic Typic Hapludalfs	No-till (corn/soybean rotation)	Spring	~8	30	38.3	1150
				Chisel plow to 200 mm	Spring	~8	30	60.3	1810
				Mouldboard ploughing to 200 mm	Spring	~8	30	31.0	930
				No-till (corn/soybean rotation)	Fall	~8	30	27.3	820
				Chisel plow to 200 mm	Fall	~8	30	40.3	1210
				Mouldboard ploughing to 200 mm	Fall	~8	30	21.7	650
Reicosky & Lindstrom (1993)	Minnesota (USA)	Canopy chamber	Clay loam or silty clay loam\	Mouldboard ploughing to 200 mm		9	19	131.1	2490
				Chisel plow to 200 mm		9	19	52.6	1000
				No-till		9	19	26.3	500
Chatsikh <i>et al.</i> (2008)	Denmark	Dynamic chamber	Typic Hapudult	Direct drilling		~53		49 <sup>a</sup>	
				Rotary harrow to 50-100 mm depth		~53		52 <sup>a</sup>	
				Rotary harrow to 50-100 mm and then		~53		60 <sup>a</sup>	
				Mouldboard plough to 200 mm		~53			

<sup>a</sup>Average from measurements during about 2 months in autumn and 3 months in spring for 2 years.

### 2.7.2 *Pugging*

Soil pugging is a term used to describe the plastic deformation of wet soil by animal hooves (Greenwood and McKenzie, 2001). Poaching and treading damage are other terms used to describe this process. Pugging usually occurs during winter when soils are wet and stocking intensities are highest (e.g. strip grazing). On dairy farms, pugging is likely to occur across much of the farm during a number of years (Singleton and Addison, 1999).

Pugging causes significant disturbance of the surface soil (like cultivation) and also destroys soil aggregates (Scholefield and Hall, 1985) and hence could enhance microbial decomposition of soil C. Pugging also damages plant roots and reduces pasture production (Edmond, 1963; Zegwaard, 2005), meaning that C inputs to the soil are likely to be reduced. However, pugging also causes a reduction in soil macro-porosity and hydraulic conductivity (Nguyen *et al.*, 1998; Zegwaard, 2005) and an increase in soil moisture content (Menneer *et al.*, 2005a) which may decrease oxygen availability and thus decrease microbial activity (Davidson *et al.*, 2000). Consequently, whether pugging would increase or decrease CO<sub>2</sub> production and emissions is unclear and there has been very little research on this topic.

#### 2.7.2.1 *Effect on soil C*

No studies were found specifically investigating the effect pugging has on soil C content, although Ferrero (1991) carried out simulated cattle pugging using a pneumatic compactor in an agroforestry area in Italy. Organic matter concentration in the top 100 mm of soil was significantly reduced under the most severely compacted treatment compared to the control. No explanation of the mechanisms thought to cause this C loss was given by Ferrero (1991). There has, however, been considerable research into the effect that pugging has on pasture production (Edmond, 1963; Drewry and Paton, 2005; Menneer *et al.*, 2005b; Zegwaard, 2005), with most studies reporting a large reduction in above-ground pasture production following pugging damage (Table 2.4). This reduction in above-ground pasture production would probably also lead to a decrease in below ground root production and therefore C inputs to the soil, which may in turn lead to a decrease in soil C.

### 2.7.2.2 Effect on soil CO<sub>2</sub> emissions

Only one study was found where CO<sub>2</sub> emissions from pugged soil had been measured, in a cattle overwintering area in the Czech Republic (Hynst *et al.*, 2007). Hynst *et al.* (2007) measured CO<sub>2</sub> emissions during about a two-year period at three points along a gradient of cattle impact (from high impact near the wintering barn to low impact a distance away from the barn). In the highly impacted area, cattle treading meant the soil was virtually devoid of plants and the surface soil severely damaged, in the moderately impacted area soil and vegetation were only moderately damaged, and in the control site, little or no impact was noted. They found that CO<sub>2</sub>-C emissions averaged 20 kg C ha<sup>-1</sup> day<sup>-1</sup> (average over two years) in the most highly impacted area near the wintering barn, 26.2 kg C ha<sup>-1</sup> day<sup>-1</sup> in the moderately impacted area and 42 kg C ha<sup>-1</sup> day<sup>-1</sup> from the control area. Hynst *et al.* (2007) suggested that reduced aeration due to damaged soil structure was one possible cause for the decreased emissions from the impacted areas. Less root respiration from the impacted areas was another suggested cause for the lower emissions from these two areas compared to the control.

A few studies have investigated the effect that compaction caused by machinery has on gaseous emissions from pastoral and cropping soils and these studies could provide an insight into the possible impacts that pugging might have on CO<sub>2</sub> emissions. Yamulki and Jarvis (2002) compared CO<sub>2</sub> and CH<sub>4</sub> emissions (over 21 days) from uncompacted soils (pasture and tilled) with those compacted using a roller. Methane emissions were over four times higher in compacted plots than the control and CO<sub>2</sub> emissions also increased although the difference was not significant. The increase in CH<sub>4</sub> emissions may have been caused by an increasingly anaerobic environment following compaction, while the increase in CO<sub>2</sub> emissions may have been due to increased microbial respiration as a result of more substrate becoming available if aggregates were crushed (Grandy and Robertson, 2007). On the other hand, Jensen *et al.* (1996) found that CO<sub>2</sub> emissions decreased significantly, immediately after a tractor compacted pastoral and tilled soils. Jensen *et al.* (1996) attributed the reduction to decreased aeration, although they noted that there may not have been a decrease in soil respiration, as measurements were made immediately after compaction, which meant CO<sub>2</sub> levels



may have just been building up in the soil. Longer term studies would be required to gain a fuller understanding of the effect of compaction on CO<sub>2</sub> and CH<sub>4</sub> emissions. Pugging does not always cause compaction, for example, when soils are pugged at high moisture contents (Zegwaard, 2005). However, in this situation aggregate breakdown still occurs (Scholefield and Hall, 1985), which could increase microbial respiration due to increased substrate availability (Grandy and Robertson, 2007).

**Table 2.4:** The effect of pugging on pasture production in studies carried out in New Zealand and Australia.

Reference	Stock class	Management		Time since pugging	Cumulative kg DM ha <sup>-1</sup>		% reduction in DM production	Comments
		Control	Treatment		Control	Treatment		
Zegwaard (2005)	Dairy cows	No pugging	Pugged 24 hours, 300 cows ha <sup>-1</sup> , 65% GSM*	8 weeks	1652	789	-52	Not significantly different after 14 weeks
	Dairy cows	No pugging	Pugged 24 hours, 300 cows ha <sup>-1</sup> , 71% GSM*	9 weeks	1279	626	-51	Not significantly different after 18 weeks
	Dairy cows	No pugging	Pugged 24 hours, 300 cows ha <sup>-1</sup> , 81% GSM*	10 weeks	1931	936	-52	Not significantly different after 22 weeks
Menneer <i>et al.</i> (2005b)	Dairy cows	No pugging	Moderate pugging (single event)	12 months	10149	8562	-16	Clover production was reduced more than ryegrass production
	Dairy cows	No pugging	Severe pugging (single event)	12 months	10149	6683	-34	
Nie <i>et al.</i> (2001)	Dairy cows	No pugging	Light pugging (single event)	4 months	9366	8717	-7	Not significantly different
	Dairy cows	No pugging	Severe pugging (single event)	4 months	9366	5632	-40	Tiller density showed similar decrease
	Dairy cows	No pugging	Single severe pugging event, Allophanic soil	2 months			-32**	Recovered after about 4 months
Ledgard <i>et al.</i> (1996)	Dairy cows	No pugging	Single severe pugging event, Gley soil (Te Kowhai)	2 months			-45**	Recovered after about 8 months
	Dairy cows	No pugging	Single severe pugging event, Gley soil (Hauraki)	2 months			-80**	Recovered after about 8 months
Drewry and Paton (2005)	Sheep	Not grazed during winter	One pugging event in winter 1800 sheep ha <sup>-1</sup>	Growth over spring	4268	6000	41	Not significantly different over the full year
	Sheep	Not grazed during winter	One pugging event in winter 1800 sheep ha <sup>-1</sup>	Growth over autumn	5639	4792	-15	
Edmond (1963)	Sheep	No pugging	One severe pugging event in winter on saturated soil	5 months	2224	303	-86	
	Sheep	No pugging	One severe pugging event in summer on saturated soil	3 months	2561	258	-90	
	Sheep	No pugging	One severe pugging event in winter on soil at field capacity	5 months	2224	1876	-16	
	Sheep	No pugging	One severe pugging event in summer on soil at field capacity	3 months	2561	1213	-53	

\*GSM = gravimetric soil moisture content. \*\* Approximate, as values were read off a graph.

## 2.8 Summary and conclusions

The C balance of New Zealand pastoral soils is poorly understood. Studies that have directly measured changes in soil C have reported mixed results, with some studies showing increases in soil C (Nguyen and Goh, 1990; McIntosh *et al.*, 1999), some no change (Tate *et al.*, 1997) and some decreases (Lambert *et al.*, 2000; Schipper *et al.*, 2007). Direct measurement of changes in soil is constrained by the long time periods required to detect changes and when using this approach factors driving changes in soil C are usually hard to identify.

The magnitude of the different inputs and outputs of C from pastoral soils is also poorly understood. Carbon inputs derived from on site photosynthesis are usually the largest C input to pasture soils. While above ground net primary production (NPP) for intensively grazed New Zealand pastures is reasonably well quantified, total C input via photosynthesis (gross primary production (GPP)) is not. This is because it is difficult to directly measure GPP and calculating GPP from above ground NPP requires assumptions about below ground NPP and autotrophic respiration (AR) ( $GPP = NPP + AR$ ). The amount of C imported in supplementary feed can vary considerably between farms and is not well documented in the literature.

Respiration is the largest output of C from pastoral soils, but there are few annual measurements of respiration from pastoral soils in New Zealand. Carbon export in milk is usually the next largest C loss from dairy farm systems and can be quantified reasonably accurately from farm production records. Leaching losses of C is one of the most poorly understood components of the pastoral C balance. There are very few published studies of C leaching losses from pastoral soils, with reported losses varying considerably, depending on soil type and drainage. Further research into C leaching losses from pastoral soils is required if this gap in knowledge is to be filled. In contrast, a number of studies have reported rates of methane emissions from grazing ruminants. However, many of these studies are not published in peer reviewed journals and a published review of methane measurements seems warranted. Most of New Zealand's intensively grazed pastures are on flat or rolling land, where C loss via erosion is generally low.

The eddy covariance (EC) technique continuously measures C exchange between the land surface and atmosphere (the balance between respiration and photosynthesis which are the largest fluxes of C in pastoral systems) and therefore may help improve our understanding of the C balance (and factors controlling the C balance) of New Zealand's pasture soils. EC studies are most useful when all other flows of C to and from a farm system are quantified, so a full farm scale C budget can be calculated. However, only one such EC study has been conducted in New Zealand (Nieveen *et al.*, 2005) and this was on a drained peat soil which is relatively un-representative of many New Zealand pasture soils. Globally there are also few EC studies of intensively grazed pasture systems (particularly studies accounting for all inputs and outputs) and New Zealand pastures are managed differently from most overseas pastures with year round grazing. Therefore, the C balance of New Zealand pastoral systems is unlikely to be able to be predicted from international data and more measurements from New Zealand systems are required.

In addition to calculating an annual farm scale C balance, quantifying the effect that specific management practices have on soil C levels will further improve understanding of the soil C balance of New Zealand pasture systems. Physical disturbance (such as cultivation) is well documented to cause C loss from soils and cultivation and cattle pugging are two types of physical disturbance that occur in New Zealand pastoral systems. As dairy farms continue to intensify, the frequency of cultivation is likely to increase as farmers increase the rate of pasture renewal and sow crops to boost total dry matter production (Clark *et al.*, 2007). The effect that occasional cultivation has on soil C is poorly understood and clearly further information is required in this area. Soil pugging by cattle hooves is relatively common on dairy farms, but there is currently very little information on the effect that pugging has on soil C and CO<sub>2</sub> emissions.

### 3. ANNUAL CARBON BALANCE OF AN INTENSIVELY GRAZED TEMPERATE PASTURE: INFLUENCE OF DROUGHT AND PHYSICAL DISTURBANCE

#### *Abstract*

Recent research showed that during the 17 – 30 years prior to 2005, soils at 31 flat or gently rolling pastoral sites (mainly dairy farms) throughout New Zealand lost on average  $1,060 \text{ kg C ha}^{-1} \text{ y}^{-1}$ . These carbon (C) losses are of concern due to the importance of C for maintaining soil quality and minimising New Zealand's greenhouse gas footprint. The cause(s) of the C loss from these soils was not known. The main objective of this research was to determine an annual farm scale C budget for an intensively grazed dairy farm. A second objective was to quantify the  $\text{CO}_2$ -C losses associated with two forms of soil physical disturbance that occur on dairy farms: cultivation for pasture renewal, and soil pugging by dairy cattle.

Net ecosystem exchange (NEE) of  $\text{CO}_2$  was measured using an eddy covariance (EC) system from 15 December 2007 to 14 December 2008. Closed chamber techniques were used to measure  $\text{CO}_2$  emissions from three cultivated paddocks and three adjacent pasture paddocks between 26 January 2008 and 5 March 2008.  $\text{CO}_2$  emissions were also measured using chambers from pugged and control plots between 25 June and 5 August. Coincidentally this research was carried out in a year with an extreme drought and a wetter than usual winter.

Annual NEE measured with the eddy covariance system was  $-1,843 \text{ kg C ha}^{-1}$ . Accounting for C in supplement import, milk export, pasture export and losses in methane, the dairy pasture system was a net sink of  $-880 \pm 500 \text{ kg C ha}^{-1}$ . This C sequestration occurred despite a severe drought during the study, which was in contrast to other studies of grasslands during drought.

Cultivation under dry conditions did not increase cumulative  $\text{CO}_2$ -C emissions compared to adjacent pasture paddocks. However, when C inputs to pasture paddocks via photosynthesis were included in calculations, net C loss from the cultivated paddocks was estimated to be  $622 \text{ kg ha}^{-1}$  more than the pasture paddocks, during the 39 day study.  $\text{CO}_2$  emissions were lower from pugged plots compared to control plots, probably caused by the combination of decreased microbial respiration and root respiration due to wetter soil conditions and lowered root respiration, as a result of lower pasture production.

These results suggest that management related soil disturbances of occasional cultivation for pasture renewal and soil pugging, are unlikely to cause large losses of soil C. Further, a severe drought also did not cause  $\text{CO}_2$ -C losses from the land surface to the atmosphere on an annual scale, in contrast to previous studies.

**Keywords:** Eddy covariance, chamber, cultivation, pugging, carbon dioxide, soil carbon.

### 3.1 Introduction

Soils contain the largest pool of terrestrial carbon and even small changes in soil C content could have a large effect on the global C budget (Tate *et al.*, 1997; Amundson, 2001). Carbon is also important for soil quality, and adequate soil C is essential to maintain the productive capacity of terrestrial ecosystems (Conant *et al.*, 2001). Changes in land use and management are well known to cause changes in soil C content. Conversion from forest to pasture results in either no change (Murty *et al.*, 2002) or a slight increase in soil C (Guo and Gifford, 2002), while conversion of forest or pasture to long term cultivation generally causes a large loss of soil C (Davidson and Ackerman, 1993; Haynes and Tregurtha, 1999; Shepherd *et al.*, 2001). Converting cultivated land to pasture almost always results in an increase in soil C (Shepherd *et al.*, 2001; Guo and Gifford, 2002). Much less is known about the effects of land use intensification on soil C stocks. Intensity of pastoral land use has increased globally to meet increasing food demands (Steinfeld *et al.*, 2006). Farm systems in New Zealand have also intensified, especially dairy farms (MacLeod and Moller, 2006). Between 1992 and 2007 milksolid production (milk fat + milk protein) on New Zealand dairy farms increased by 43% (Livestock Improvement, 2007) due to increased use of nitrogen fertilizer, better pasture utilization, more supplementary feeding (Clark *et al.*, 2007) and higher stocking rates (Parliamentary Commissioner for the Environment, 2004).

Until recently, it was generally thought that soil C levels under established pastoral agriculture in New Zealand were at near steady state (Tate *et al.*, 1997; Tate *et al.*, 2005). However, some studies have shown C losses from soils under pastoral agriculture. Lambert *et al.* (2000) found that a hill soil under pasture in New Zealand lost  $200 \text{ kg C ha}^{-1} \text{ y}^{-1}$  between 1972 and 1987 and Schipper *et al.* (2007) demonstrated that during the 17 – 30 years prior to 2005, soils at 31 flat or gently rolling pastoral sites throughout New Zealand lost on average  $1,060 \text{ kg C ha}^{-1} \text{ y}^{-1}$ . The losses observed by Schipper *et al.* (2007) were similar to losses reported by Bellamy *et al.* (2005) for soils under pasture in England and Wales.

The cause(s) of large C losses from soils under pastoral agriculture remain unclear. Change in soil C content occurs when there is an imbalance between C

inputs (photosynthesis or feed imports) and C outputs (respiration, leaching, erosion and product) (Amundson, 2001). In pastoral ecosystems, the soil C balance is dominated by photosynthesis and respiration (Post *et al.*, 2001; Byrne and Kiely, 2006). Changes in environmental conditions or management practices can affect the magnitude of photosynthesis and respiration, because both are strongly influenced by a range of physical, chemical and biological factors (Post *et al.*, 2001). For example, drought causes a net loss of C from the land surface to the atmosphere, because during drought, photosynthesis decreases proportionally more than respiration (Flanagan *et al.*, 2002; Aires *et al.*, 2008; Baldocchi, 2008). Grazing events (Nieveen *et al.*, 2005) and harvesting pasture for silage or hay (Jaksic *et al.*, 2006) have also been shown to cause short term net loss of C from the land surface to the atmosphere.

The most well-documented cause of C loss from soils is physical disturbance, such as continuous cultivation in cropping systems (Davidson and Ackerman, 1993; Haynes and Tregurtha, 1999; Shepherd *et al.*, 2001). Physical disturbance tends to destroy soil aggregates, exposing previously protected C to microbial decomposition and increasing soil respiration (Tisdall and Oades, 1982; Six *et al.*, 2004; Grandy and Robertson, 2007). In contrast to arable or horticultural land, cultivation of pastoral land is infrequent, generally only being used to establish forage crops and sow new pasture. However, the largest losses of soil C generally occur in the first few years after cultivation (Davidson and Ackerman, 1993; Haynes and Tregurtha, 1999), and therefore periodic cultivation could contribute to a loss of soil C if there was not sufficient time between cultivation events for C levels to be replenished (Martel and Mackenzie, 1980). There is a paucity of information about the effect of occasional cultivation on soil C content, particularly in pastoral systems (Conant *et al.*, 2007).

Pugging (the plastic deformation of wet soil by animal hooves, Greenwood and McKenzie, 2001) is another form of physical disturbance which can occur in pastoral systems (particularly intensive systems). Like cultivation, pugging also destroys soil aggregates (Scholefield and Hall, 1985) and hence could also enhance microbial decomposition of soil C. Pugging also damages plant roots and reduces pasture production (Edmond, 1963; Zegwaard, 2005), meaning that C

inputs to the soil are likely to be reduced. On intensive dairy farms in New Zealand, Singleton and Addison (1999) suggested that over a period of a few years pugging could occur over much of a farm. The effects of cattle pugging on soil physical properties (Drewry and Paton, 2000; Zegwaard, 2005) and pasture production (Edmond, 1963; Menneer *et al.*, 2005b) has been well documented. Studies have also examined the effect that pugging has on nutrient and sediment losses (Nguyen *et al.*, 1998; McDowell *et al.*, 2003), earthworm abundance (Edmond, 1963; Drewry and Paton, 2005) nitrous oxide emissions (Thomas *et al.*, 2008) and denitrification (Menneer *et al.*, 2005a). No work, however, has specifically examined the effect that pugging has on soil CO<sub>2</sub> emissions and the consequent impact on soil C content.

Determining the cause(s) of C changes in pastoral soils is methodologically difficult. Over short time periods (e.g. <10 years), the magnitude of any C loss or gain from soil is usually small (generally 0 – 10,000 kg C ha<sup>-1</sup> y<sup>-1</sup>) in relation to the large background C content (e.g. NZ average is 168,000 kg C ha<sup>-1</sup> in the top 1 m of soil, Tate, *et al.*, 1997) and spatial variation in soil C content is usually large (Brye *et al.*, 2002). These two factors make measuring changes in soil C content directly over relatively short-time periods difficult (Brye *et al.*, 2002; Conen *et al.*, 2003).

Another approach commonly used to study the carbon balance of terrestrial ecosystems is the micrometeorological technique—eddy covariance (EC) (Baldocchi, 2008). EC systems continuously measure the exchange rate of CO<sub>2</sub> between the land surface and atmosphere at hectare scales by measuring the covariance between vertical wind speed and CO<sub>2</sub> concentrations, Baldocchi 2003). The large scale approach of EC makes it well suited to measurement at the paddock scale on farms. There have been a number of EC studies over pasture (Gilmanov *et al.*, 2007; Soussana *et al.*, 2007), but relatively few on intensive pastures that are grazed year round, such as those common in New Zealand (except see, Nieveen *et al.*, 2005). EC studies can be coupled to chamber measurements of CO<sub>2</sub> flux in smaller experimental treatment plots, to measure effects of specific management practices (e.g. grazing or cultivation) (Rochette and Hutchinson, 2005).



The main objective of this research was to determine an annual farm scale C budget for a dairy farm (Waikato, New Zealand), by combining data from CO<sub>2</sub>-C exchange measurements made with an eddy covariance system, with other fluxes of C, calculated from farm production data and literature values. A second objective was to determine the magnitude of the CO<sub>2</sub>-C loss associated with two forms of physical disturbance of a dairy farm soil: cultivation for pasture renewal, and soil pugging by dairy cattle (using closed chamber techniques).

There was a severe drought during the summer/autumn of 2008 with rainfall in January being the lowest in the last 100 years. This provided an opportunity to observe the extent to which drought affected seasonal and annual C budgets.

## 3.2 Methods

### 3.2.1 *General site description*

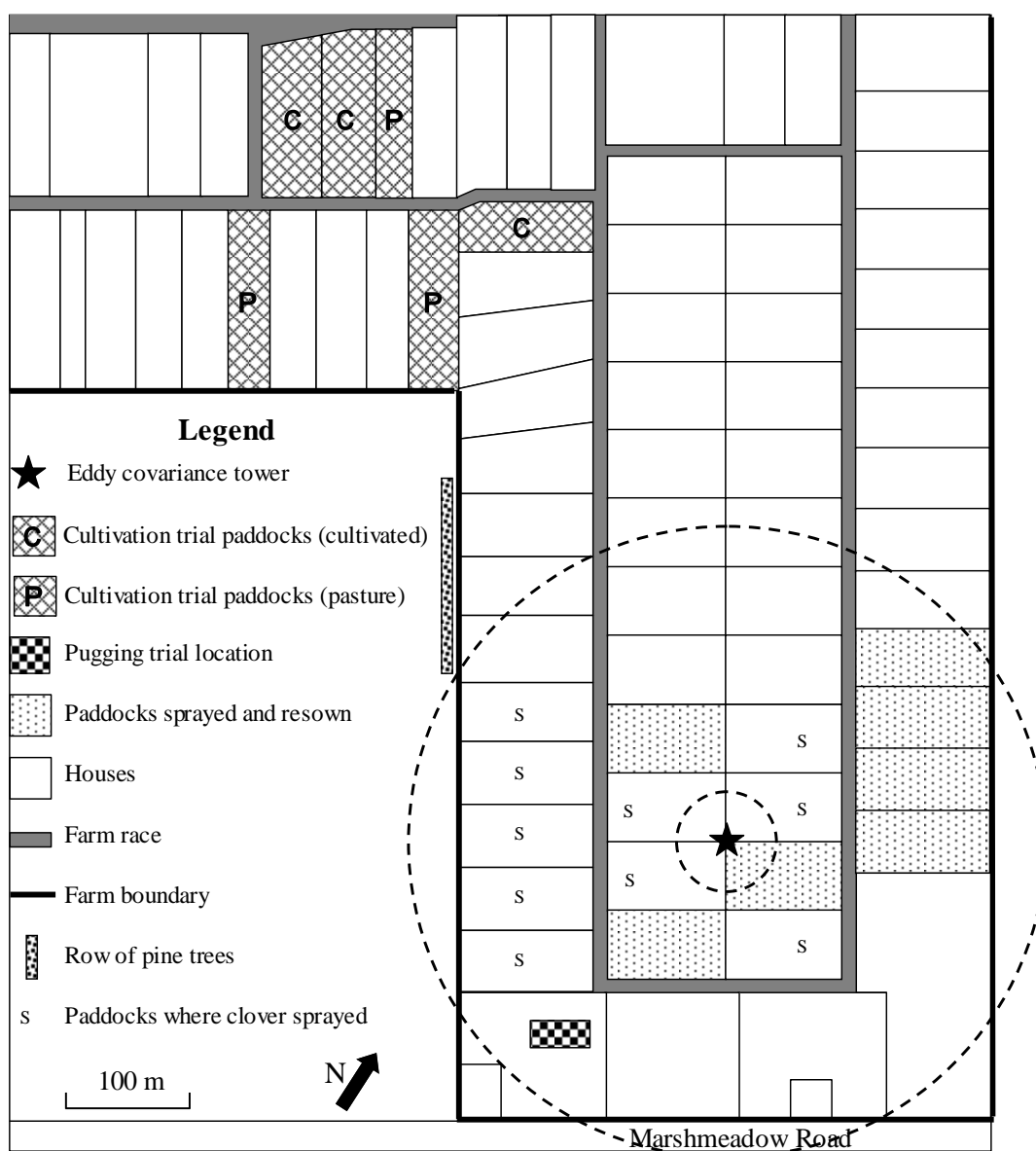
This study was conducted at a research dairy farm (Scott Farm, owned and operated by DairyNZ) located 7 km northeast of Hamilton in the Waikato, North Island, New Zealand (37.46° S 175.22° E). Scott Farm is an intensive dairy farm where DairyNZ conducts large scale farm systems trials. Cattle are grazed outdoors on pasture all year round. The nearest climate station to Scott Farm was at Ruakura (about 6 km to the northwest) where 30 year (1978 – 2007) mean annual rainfall and temperature were 1126 mm and 13.8 °C respectively (NIWA, n.d.). A range of soil types are present on Scott Farm but this research was conducted on an extensive area of the Matangi silt loam, (Typic Orthic Gley Soil, Hewitt, 1993). The surface horizon (0 – 200 mm) and upper subsoil had loamy silt textures, while the lower subsoil (>600 mm) contained large amounts of coarse sand and fine and coarse gravel (Stiles, 1998). Total C and N in the surface soil (0 – 100 mm) were 7.7% and 0.72% respectively.

### 3.2.2 Eddy covariance study

#### 3.2.2.1 Site description

An EC tower was established in an open pasture area with the distance to the farm boundary being about 200 m to the east and west, 250 m to the south and >500 m to the north (Fig.3.1). The EC tower was surrounded by a number of small farmlet trials, with paddocks rotationally grazed by small herds of dairy cows (9 – 21 cows herd, overall farmlet stocking rates  $\sim 3$  cows  $\text{ha}^{-1}$ ). The small herd size meant that at any one time, paddocks within the flux footprint had a range of pasture biomasses. Between 20 January and 30 January 2008, grass was harvested (cut to 40 mm) from a number of paddocks within the flux footprint and removed from the farm to be fed in an indoor feeding trial. Due to a severe summer drought, palm kernel extract (PKE) was brought into the farm system to supplement cow diets in autumn and winter. Both supplement imports and exports were quantified by DairyNZ staff.

At the start of the study, pastures in all paddocks within the flux footprint were predominantly comprised of perennial ryegrass (*Lolium perene*) and white clover (*Trifolium repens*). On 16 April 2008, seven paddocks within the flux footprint were sprayed with herbicide to kill existing pastures, and on 28 April these paddocks were re-sown with perennial ryegrass seed (direct drilled using a cross slot drill) (Fig. 3.1). Between 24 April and 13 May, white clover was removed from the pasture in all paddocks immediately surrounding the EC tower (Fig. 3.1 1), by spraying with Dicamba herbicide ( $2 \text{ l ha}^{-1}$ ). Spraying was repeated on 20 June. Pasture growth rates and total pasture accumulation (above 20 mm) was determined for paddocks around the EC tower by DairyNZ staff using the “rate of growth” cutting technique similar to that described by Radcliffe (1974). Two cages were placed in each paddock and pasture within the cages was trimmed to 20 mm (minimum grazing height). Cages were then left for about 20 – 50 days (similar to grazing intervals) and then a  $0.2 \text{ m}^2$  quadrat was cut to 20 mm. Cages were then moved to a new area of the paddock, trimmed to 20 mm again and the same process repeated throughout the year. During the study period approximately  $220 \text{ kg ha}^{-1}$  of nitrogen was applied in the form of urea (all applied from April 2008 – December 2008).



**Fig. 3.1.** Location of the three experimental sites on Scott Farm along with other farm features (see legend). Also shown are paddocks around the EC tower which were sprayed with herbicide and re-sown with ryegrass seed, paddocks in which clover was sprayed, and the average distance to the peak of the daytime flux footprint (xMAX, inner circle) and the distance within which 80% of daytime fluxes were sourced (CNF80, outer circle) (see text for details). Adapted from DairyNZ Scott Farm map.

### 3.2.2.2 Instrumentation

The EC system comprised of a 3D sonic anemometer (CSAT3, Campbell Scientific Inc, Logan, UT, USA) and an open path IRGA (model LI-7500, LICOR Inc., Lincoln, NE, USA) attached to a steel tower (400 mm lattice triangle) at 2.84 m. The IRGA was installed horizontally to minimise the time water droplets were present on the IRGA windows and was calibrated prior to installation in the field in December 2007, and subsequently on 5 March, 26

August and 17 December 2008. Between 5<sup>th</sup> and 27<sup>th</sup> March, the CSAT3 sonic anemometer was factory re-calibrated. During this time, a RM YOUNG sonic anemometer (RMY8100) was used to measure 3D wind speed and sonic temperature.

Air temperature and humidity (HMP 45A, Vaisala, Finland) were also measured at 2.84 m, while wind direction (W200P, Vector Instruments, Clwyd, UK), Photosynthetic Photon Flux Density (PPFD) (LI-190SA, LI-COR, Inc.) and global radiation (R<sub>g</sub>) (LI190SB, LICOR Inc) were measured at 3 m. A cup anemometer (A101M, Vector Instruments, Clwyd, UK) and a net radiometer (Q6.7.1, Radiation and Energy Balance Systems, Seattle, WA, USA) were mounted at 0.93 m. Precipitation was measured with a tipping bucket rain gauge (TB5, Hydrological Services) at 0.4 m. Soil temperature at 50 mm was measured using a four junction averaging thermocouple (TCAV, Campbell Scientific Inc.) and at 100 mm with a soil thermister. Volumetric soil moisture content was measured using two water content reflectometers (CS616, Campbell Scientific Inc.) at 50 and 100 mm depth, calibrated for this soil (Appendix A). Instruments were powered by 12 V batteries (R220, ENDURANT) which were kept charged by four solar panels (SX-80U, BP Solar).

### ***3.2.2.3 Data acquisition and initial processing***

A CR3000 (Campbell Scientific Inc) data logger sampled signals (at 20 Hz) from the IRGA and sonic anemometer and raw signals were stored on a compact flash card. Fluxes of CO<sub>2</sub> and energy were calculated online and stored as 30 minute averages. These data were corrected following the methods of Nieveen *et al.* (2005), which included density corrections (Webb *et al.*, 1980), co-ordinate rotation corrections (McMillen, 1988), sonic temperature corrections (Schotanus *et al.*, 1983) and frequency response corrections (Moore, 1986). These corrections were made using a custom MATLAB 7.5 program (The Mathworks Inc., Natick, MA, USA) (Nieveen *et al.*, 2005). Measurements of PPFD, net radiation, air and soil temperature, humidity and precipitation were also sampled at 20 Hz by the CR3000 data logger and stored as 30 minute averages (except precipitation which was a 30 minute total). Data recorded by the CR3000 data logger were backed up daily via an automated telemetry system. A CR10X data logger (Campbell

Scientific Inc) recorded all remaining variables which were stored as 30 minute averages and manually downloaded approximately every two weeks.

#### **3.2.2.4 Data quality control and gap filling**

During rain, frost and fog, liquid water or ice on or between the open path IRGA windows affected IRGA readings. These corrupted data were identified by comparing vapour pressure measured by the IRGA with that measured by the HMP 45A, which was un-affected by rain. If the vapour pressure measured by the IRGA deviated by more than 30% from that measured by the HMP 45A sensor data were rejected (Rogiers *et al.*, 2005). At night, when friction velocity ( $u_*$ ) was below  $0.1 \text{ m s}^{-1}$ , net ecosystem  $\text{CO}_2$  exchange (NEE) decreased and therefore data below this threshold were eliminated (Smith, 2003; Rogiers *et al.*, 2005; Pereira *et al.*, 2007). After this filtering process there were a few remaining values outside the natural range and these were eliminated based on simple threshold values ( $>25$  and  $< -40 \text{ } \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ ). After filtering there was 53% good data remaining, which was lower than the average of 65% reported by Falge *et al.*, (2001), largely due to low windspeeds at this inland basin site and a large number of periods with rain. Gaps were filled and NEE partitioned between gross primary productivity (GPP) and total ecosystem respiration (TER) using the online software described by Reichstein *et al.* (2005) (see Appendix B).  $\text{CO}_2$  storage corrections were not applied in this study, but on a nearby pasture site, Nieveen *et al.* (2005) found that storage was negligible, as did Smith (2003), who did a more comprehensive study on  $\text{CO}_2$  storage in a nearby peat bog.

#### **3.2.2.5 Footprint analysis**

The size of the footprint from which EC measurements was determined following the method of Schuepp *et al.* (1990). Footprint analysis was only carried out for half hour periods with valid NEE data. For measurements during daylight, the distance to the peak of the flux footprint (xMAX) was on average 26 m compared to 50 m at night. Adequate fetch is often defined as the distance within which 80% of flux measurements are sourced (CNF80) (Thompson *et al.*, 1999; Gockede *et al.*, 2008). On average, 80% of fluxes during the day were sourced from within a 235 m radius of the tower, while at night 80% of fluxes were sourced from within a 455 m radius (Fig. 3.1). This analysis showed that during the day, fetch was

nearly adequate, but at night, fetch was too short and the source area of some flux measurements would have been outside the farm boundary. However, the major land use beyond the farm boundary was pastoral agriculture and therefore the few additional fluxes from these areas were expected to be similar to those from within the farm boundary.

### **3.2.3 *Cultivation trial***

#### **3.2.3.1 *Site description and trial design***

The cultivation trial consisted of six paddocks (3 cultivated and 3 pasture controls) on the Matangi soil, but outside the EC flux footprint (Fig. 3.1). These paddocks had previously been grazed in the same fashion as paddocks around the EC tower. The 3 cultivated paddocks were sprayed with a glyphosate-based herbicide (7 January, 2008), mouldboard ploughed to 200 mm depth (17 – 21 January), and then power harrowed and rolled (26 January). Weeds that grew (very few) were re-sprayed with herbicide (29 February) and then on 6 March paddocks were roller tilled and seed sown.

Between 20 January and 27 January, grass in the three pasture paddocks used in this study was harvested to 40 mm (the same as paddocks around the EC tower). Above-ground pasture biomass following harvest of the pasture paddocks was about 2300 kg DM ha<sup>-1</sup> and prior to cultivation above-ground pasture biomass was about 1700 kg DM ha<sup>-1</sup> (J. Siemelink, pers comm. 2008<sup>2</sup>).

#### **3.2.3.2 *Measurements***

An IRGA (LI-8100, LI-COR Inc.) combined with a survey chamber (LI-8100-103, LI-COR Inc.) and an automatic chamber (LI-8100-101, LI-COR Inc.) were used to measure CO<sub>2</sub> emissions. The survey chamber measurements was used to determine spatial variation of CO<sub>2</sub> fluxes, while automatic chamber measurements provided a near continuous measure of CO<sub>2</sub> emissions (from one location), from which diurnal trends could be determined as well as the effect of rain.

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<sup>2</sup> Personal communication with John Siemelink, Scott Farm Manager, Dairy NZ, Hamilton.

Immediately after power harrowing finished (26 January), one large plastic collar (200 mm diameter by 114 mm deep) was inserted into one of the cultivated paddocks and into an adjacent pasture paddock (30 mm protruding above ground). CO<sub>2</sub> emissions were measured every 10 minutes using the automatic chamber, which was moved on alternate days between the collar in the cultivated paddock and the collar in the pasture paddock. Additionally, 10 plastic collars (100 mm diameter by 76 mm long) were inserted (with 25 mm protruding above ground) at random locations in each of the three cultivated and three adjacent pasture paddocks (total of 60 collars). CO<sub>2</sub> emissions were measured from these collars about every two days using the survey chamber. Measurements were made at different times each day (only during daylight) to obtain CO<sub>2</sub> fluxes for a range of soil temperatures.

Soil temperature at 50 and 100 mm (8100-201, LI-COR Inc. & Ama digit ad 170<sup>th</sup>, Precision) and volumetric moisture content in the surface 120 mm depth (Hydrosense, CS620, Campbell Scientific Inc.) were recorded at each of the survey chamber collars each time CO<sub>2</sub> emissions were measured. Soil temperature and moisture content at 50 mm next to the automatic chamber collars in the cultivated and pasture paddocks were measured with 107 thermistors (Campbell Scientific Inc) and moisture content reflectometers (CS615, Campbell Scientific Inc.). These data were recorded every 10 minutes with a CR10X data logger (Campbell Scientific Inc). The CS615 moisture content reflectometers and CS620 Hydrosense probe were calibrated for the Matangi soil (see Appendix A).

Total C loss during the cultivation study was calculated by interpolating between daily mean fluxes from survey chamber measurements. The reason survey chamber measurements were used to calculate total C loss and not the automatic chamber measurements was because although survey measurements did not account for diurnal variation in emissions, their spatial coverage meant they were more likely to be an accurate measure of total C losses than continuous measurements at one point. Automatic chamber measurements during the cultivation trial showed that on average the difference in respiration between night and day was less than 10%.

### 3.2.4 *Pugging trial*

#### 3.2.4.1 *Site description and trial design*

The pugging trial was carried out on the Matangi soil about 250 m from the EC tower (Fig.3.1). The trial was a small plot trial arranged in a randomized block design, with four blocks and two treatments (pugged and unpugged). Each plot was 7 m by 7 m, with a 2 m buffer strip around plot margins. Prior to the pugging trial, above-ground pasture biomass was  $\sim 3000$  kg DM ha<sup>-1</sup> in all plots. The day before pugging commenced, plots were mowed to 28 mm (the same height as nearby paddocks that had just been grazed) and clippings removed. The pasture was short enough to prevent cows grazing, which ensured that pasture biomass remained the same in all plots. Half of the plots were pugged once (24 June 2008) by 3 pregnant Friesian cross Jersey dairy cows per plot ( $\sim 500$  kg, 612 cows ha<sup>-1</sup>) over a 5 hour period (Fig. 3.2). The stocking rate was about twice the normal winter stocking rate. Cows were observed for the duration of the trial and the location of any urine and dung deposits accurately recorded and these areas were excluded from any subsequent measurements.



**Fig. 3.2.** (a) Photograph showing pugging trial nearing completion on 24 June 2008 and (b) individual pugged and (c) control plot soon after cows were removed from the plots.



### 3.2.4.2 *Measurements*

CO<sub>2</sub> emissions were measured using the same instrumentation and methods described for the cultivation experiment (see above). Immediately after cows had been removed from the pugged plots, 7 plastic collars (100 mm diameter x 76 mm length) were inserted at random locations within each plot. From 25 June, CO<sub>2</sub> emissions were measured every day (except day 2) for the first 7 days, then every 2 – 3 days for 2 weeks and then weekly for 3 weeks. Survey chamber measurements were made between 11:00 and 14:30 hours. Two large collars (200 mm diameter) were inserted into one of the pugged plots and one of the control plots, from which the automatic chamber measured fluxes as described for the cultivation trial.

Soil temperature and moisture measurements (at survey and automatic chamber collars) were made in the same way as for the cultivation experiment. Percentage bare ground in each plot was determined using a point analysis method similar to that described by Radcliffe and Mountier (1964) for determining pasture species composition. A 1.1 m long frame, with 10 points each 100 mm apart, was randomly placed in each plot 10 times, with the percentage bare ground determined from the number of times points were positioned above bare ground. Bulk density was determined on 5 July (10 days after pugging) by taking 5 cores (98 mm diameter by 75 mm long) from each plot, drying at 105 °C for 48 hours and then weighing. At the end of the study, pasture was harvested to ground level from four 0.2 m<sup>2</sup> quadrats in each plot, dried at 95 °C for 48 hours and then weighed to determine total biomass. Pasture was also harvested from within the CO<sub>2</sub> measurement collars three times during the measurement period to ensure the chamber sealed properly. Grass was cut to 28 mm, placed in a paper bag and then oven-dried for 48 hours at 95 °C and then weighted to determine total biomass accumulation per collar.

At the completion of the study, all collars were pushed flush with the soil surface and soil cores removed intact. Soil was then removed from the cores, weighed, sieved through a 9 mm sieve and then mixed thoroughly. A subsample from each collar was oven dried (105 °C for 48 hours) to determine soil moisture content and bulk density was calculated. Root biomass was then determined for 4 out of 7

collars from each plot. Soil was washed through stacked sieves with mesh of 1.6 mm and 0.71 mm. Material remaining on the sieves was placed in a 10 l bucket and above ground litter and seeds that floated to the surface were removed, after which any remaining soil mineral particles were separated from the roots by decanting three times (through the 0.71 mm sieve). Roots were then dried at 95 °C for 48 hours and weighed.

As for the cultivation study, total C loss during the pugging study was calculated by interpolating between daily mean fluxes from survey chamber measurements. Measurements made using the automatic chamber between 11:00 and 14:30 were on average 10% and 3% higher from the pugged and control collars respectively, than the average daily flux determined at the same time using the survey chamber. This suggests that daily survey chamber measurements will be a slight overestimate of the actual daily flux.

### **3.2.5 Data analysis**

Paired t-tests (Microsoft Office Excel, 2007) were used to determine whether there were significant differences for the different variables (e.g. cumulative CO<sub>2</sub> emissions, bulk density) between cultivated and non-cultivated paddocks and pugged and non-pugged plots. All reported differences were significant at the 5% level unless noted otherwise. The significance of the relationship between cumulative CO<sub>2</sub> emissions from individual collars and cumulative pasture production and root biomass in the pugging study was determined using the LINEST function in Microsoft Office Excel (2007) and testing the significance of the F value with the FDIST function.

## **3.3 Results**

### **3.3.1 Climatic, soil and pasture conditions**

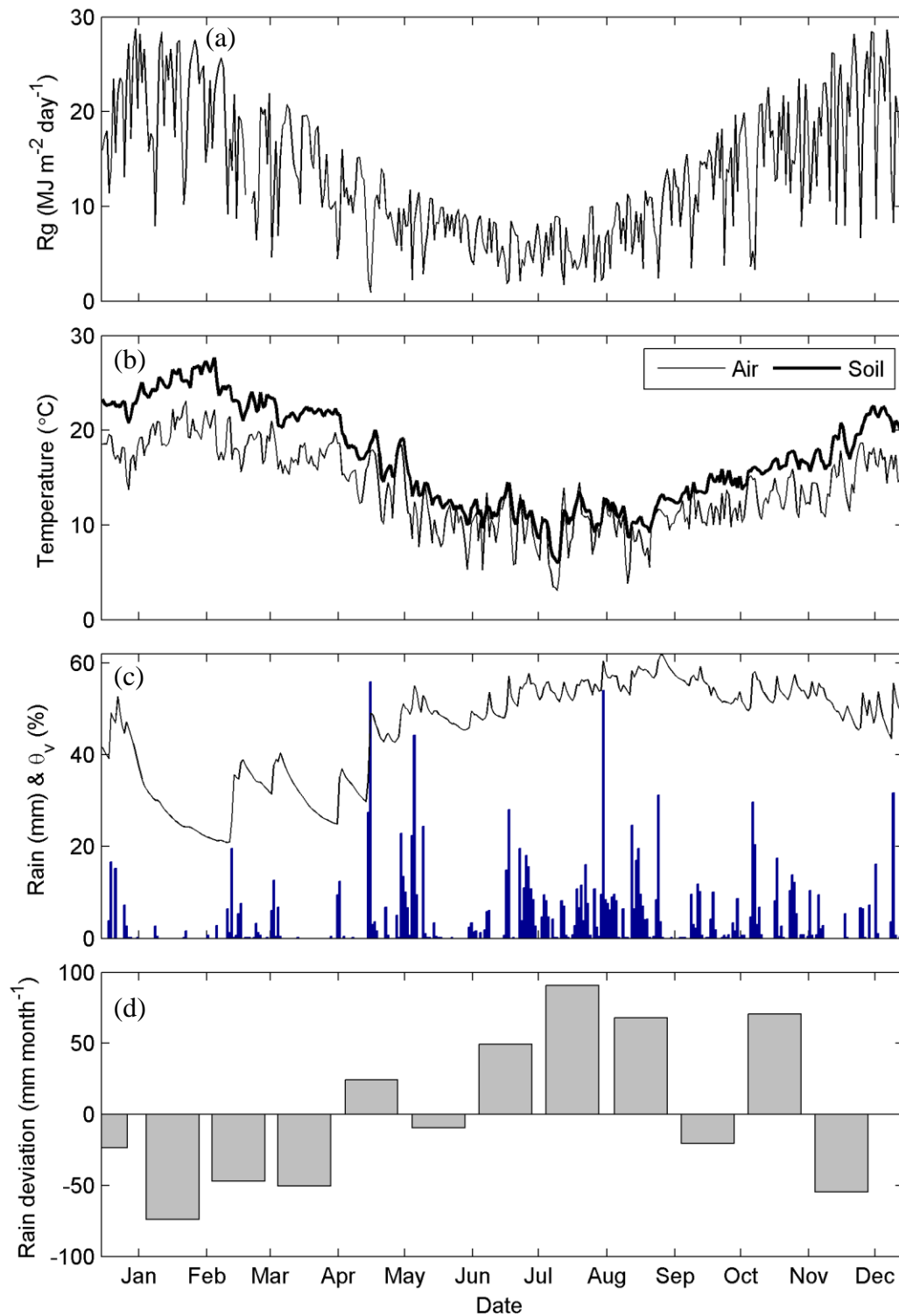
This research was carried out in a year with extreme climatic events (Fig. 3.3). Mean monthly global radiation (R<sub>g</sub>) was highest in January (21.7 MJ m<sup>-2</sup> day<sup>-1</sup>), declined gradually to July (5.5 MJ m<sup>-2</sup> day<sup>-1</sup>), before increasing again to levels similar to January by the end of the study (Fig. 3.3a). Mean monthly air

temperature at the Ruakura climate station (6 km from Scott Farm) was highest in January 20.3 °C (2 °C higher than the 30 year normal), with temperatures from December 2007 to April 2008 all being higher than average (NIWA, n.d.). Temperature in May was 1.6 °C lower than average while from June – December 2008 temperatures were almost exactly the same as the 30 year normal (NIWA, n.d.). The lowest monthly mean temperature occurred in July (9 °C). Total annual precipitation measured at the Ruakura was 1148 mm, which was similar to the 30 year normal of 1127 mm (NIWA, n.d.). However, rainfall distribution was very different to usual (Fig. 3.3d). January had the least rain since records began, and rainfall in February and March was also much lower than average, while the months of June, July and August were wetter than usual (NIWA, n.d.) (Fig. 3.3d). Volumetric soil moisture content declined rapidly from 48% on 27 December to only 21% on 9 February (Fig. 3.3c). Moderate rainfall events on 10 February, 2 March and 30 March temporarily increased soil moisture to about 40% (average of 50 and 100 mm depths) but the soil dried rapidly between these rain events. On 14 and 15 April, 83.2 mm of rain fell which increased soil moisture to about 50% and soil moisture remained above 45% until the end of the study (Fig. 3.3c).

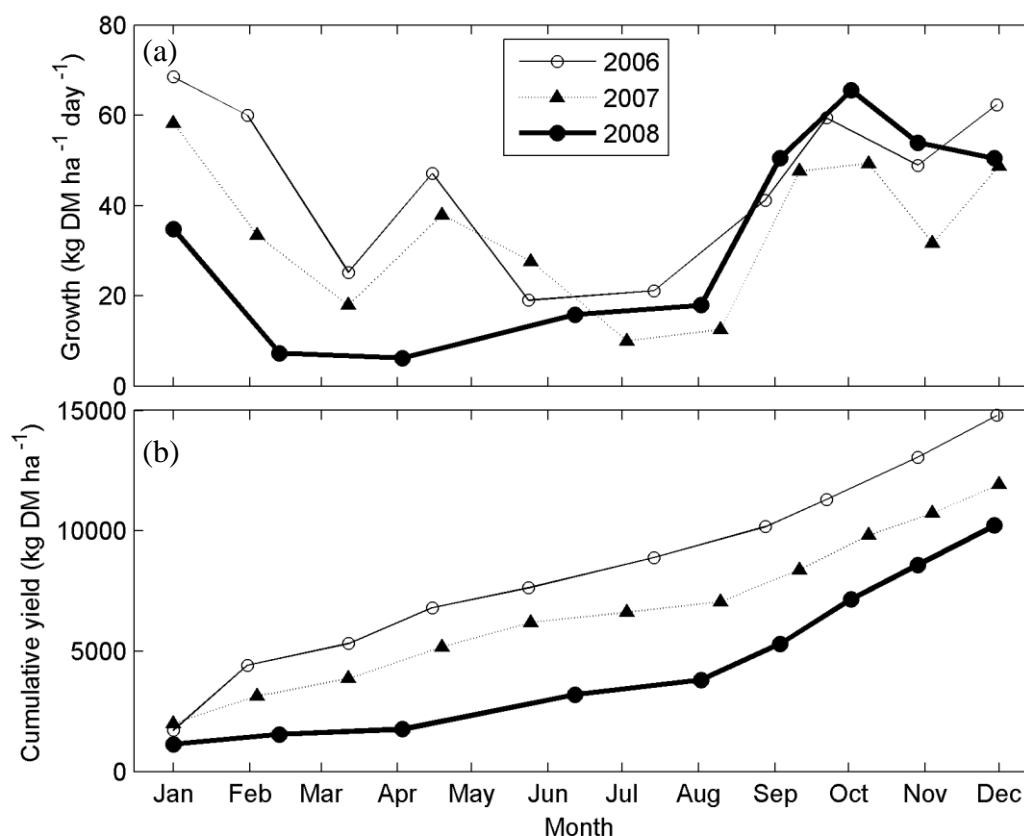
Drought conditions meant that pasture growth was much lower than usual during the summer and autumn, but growth was slightly higher during spring than the previous two years (Fig. 3.4a) (E.R. Thom, pers. comm. 2008<sup>3</sup>). Total pasture production (above 20 mm height) over the duration of the study was 10,221 kg DM ha<sup>-1</sup> y<sup>-1</sup> which was lower than in 2007 (11,934 kg DM ha<sup>-1</sup> y<sup>-1</sup>) and 2006 (14,802 kg DM ha<sup>-1</sup> y<sup>-1</sup>) (Fig. 3.4b).

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<sup>3</sup> Personal communication with Dr Errol Thom, Scientist, Dairy NZ, Hamilton.



**Fig. 3.3.** Annual trends of (a) mean daily global radiation ( $R_g$ ), (b) mean daily air temperature at 2.84 m and soil temperature at 0.1 m, (c) total daily rainfall (bars) and mean daily volumetric soil moisture content ( $\theta_v$ , line, average of 0.05 and 0.1 m depths), and (d) deviation from 30 year mean monthly rainfall (1978 – 2007).



**Fig. 3.4.** (a) Pasture growth rates determined from cage cuts around the EC tower at Scott Farm for the years of 2006, 2007 and 2008 and (b) cumulative pasture yields over the same time period. Note: both growth rate and cumulative yield values are plotted at the mean time between when the cage was first set up and when the cage was harvested (Data from DairyNZ).

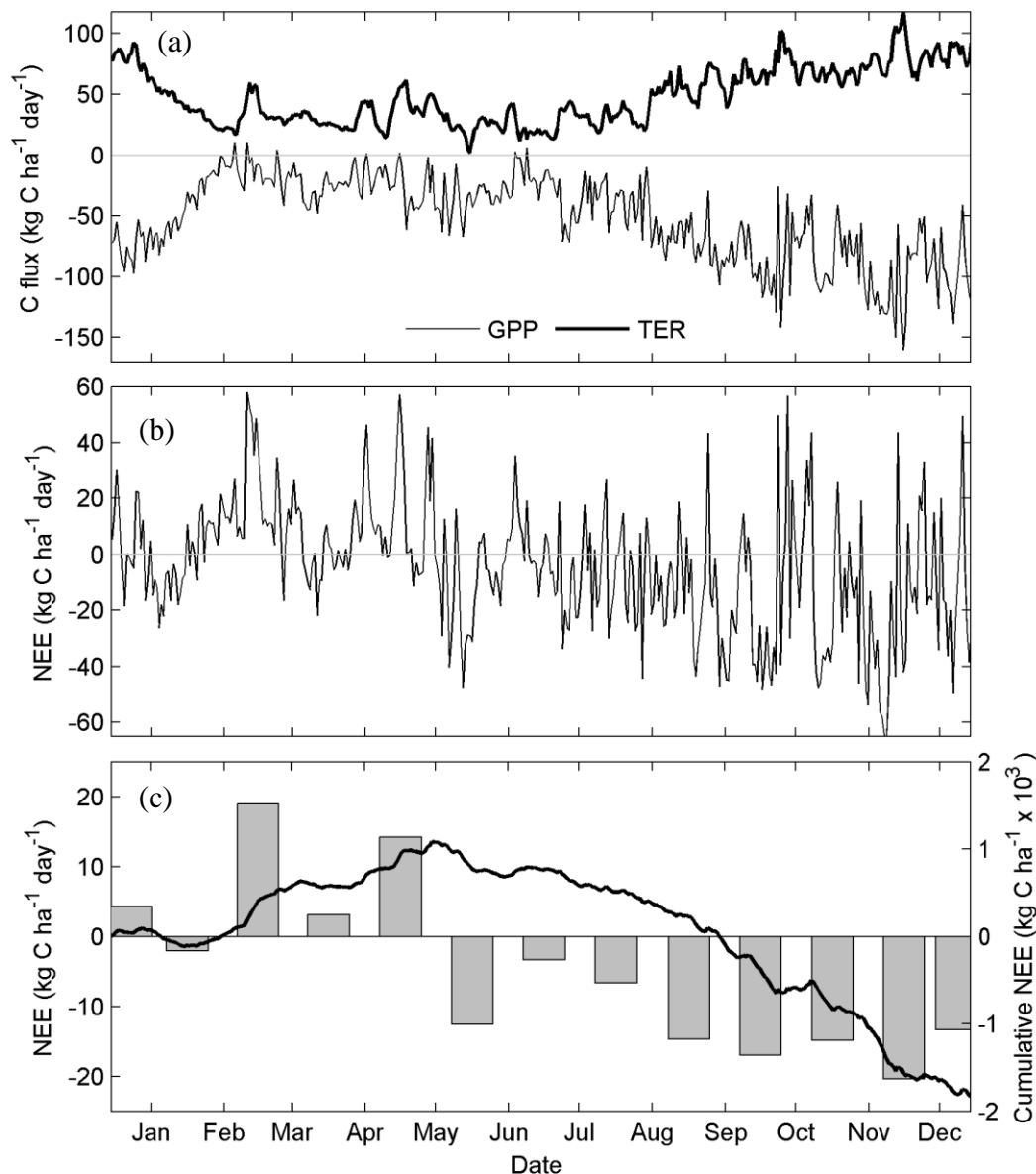
### 3.3.2 Farm scale C budget

In this paper the micrometeorological sign convention for net ecosystem exchange of CO<sub>2</sub> (NEE) was used, where gains by the land surface are negative and losses to the atmosphere are positive. For simplicity, all other inputs of C to soil-pasture system were denoted by a negative sign and all outputs as positive.

#### 3.3.2.1 Seasonal trends

Between 16 December 2007 and 9 February, total ecosystem respiration (TER) and gross primary production (GPP) decreased rapidly as the soil dried out (Fig. 3.5a). By the end of January, GPP was practically zero, TER had declined to low levels and NEE was positive indicating that the soil-pasture system had become a source of C (Fig. 3.5a, b). During December 2007, the soil-pasture system was a small net source of C while during January it was a small net sink of C (Fig. 3.5c). Rain between 10 and 17 February caused a large increase in TER, but GPP was

virtually unaffected (Fig. 3.5a). The soil-pasture system was a net C source during the months of February, March and April (Fig. 3.5c), with February having the largest C loss ( $18.9 \text{ kg C ha}^{-1} \text{ day}^{-1}$ ). In contrast to April, during May the soil-pasture system was a large net sink of C, while in June the soil-pasture system was only a small C sink. From July until the end of September, both TER and GPP gradually increased. During this period, the soil-pasture system was a net sink of C, with sink intensity increasing from June – September (Fig. 3.5c). From the end of September to 14 December, TER remained relatively constant, while GPP fluctuated considerably, but in general, remained relatively high. Maximum monthly C sequestration occurred in November ( $20.3 \text{ kg C ha}^{-1} \text{ day}^{-1}$ ).



**Fig. 3.5.** (a) Daily totals of modelled gross primary production (GPP, plotted as negative values) and total ecosystem respiration (TER), (b) daily mean net ecosystem exchange of CO<sub>2</sub> (NEE), (c) mean monthly NEE (bars) and cumulative NEE (line).

### 3.3.2.2 Annual C budget

Annual NEE measured by the EC system was  $-1,843 \text{ kg C ha}^{-1}$  (a net gain by the soil-pasture system) (Fig. 3.5c). Partitioning NEE into GPP and TER using the method of Reichstein *et al.* (2005) gave annual values of  $-19,448 \text{ kg C ha}^{-1}$  and  $17,605 \text{ kg C ha}^{-1}$  for GPP and TER respectively. A total of  $-218 \text{ kg C ha}^{-1}$  was brought into the farmlets surrounding the EC tower in palm kernel extract (PKE) (90% dry matter and assuming 50% C).

During the study period, milk fat, milk protein and lactose production from farmlets within the flux footprint averaged 550, 450 and 618  $\text{kg ha}^{-1} \text{ year}^{-1}$  respectively (E.R. Thom, pers. comm. 2008<sup>4</sup>). On average dairy cattle milk fat contains 70% C, milk protein 46% C and lactose 40% C (Wells, 2001) and therefore C export in milk was  $840 \text{ kg ha}^{-1} \text{ year}^{-1}$ . A total of  $2,967 \text{ kg ha}^{-1}$  of dry matter was harvested and exported from within the flux footprint, which equates to  $1,187 \text{ kg C}$  (assuming 40% C content, Saggar and Hedley, 2001). Dividing this by the area of the average daytime flux footprint (17 ha) gives a C export in harvested pasture of  $70 \text{ kg ha}^{-1}$ . Losses of C in methane from cows was derived from work by Laubach and Kelliher (2004) who measured methane emissions from a herd of grazing dairy cows and found that emissions averaged  $330 \text{ g CH}_4 \text{ cow}^{-1} \text{ day}^{-1}$ . Farm stocking rate at the study site was  $3.0 \text{ cows ha}^{-1}$  which means that C losses from cows via  $\text{CH}_4$  would have been about  $271 \text{ kg ha}^{-1} \text{ y}^{-1}$ . Leaching losses of C as dissolved organic carbon (DOC) and bicarbonate would have likely also occurred, but were not measured in this study. Export of C in meat would have been small because replacement heifers were grazed off farm and only brought back when mature and therefore there would have been little increase in live weight. C losses via erosion were assumed to be negligible due to the flat topography of the farm.

Summing all the inputs and outputs of C during the annual study period showed that the soil-pasture system was a sink of  $-880 \text{ kg C ha}^{-1}$  (Fig. 3.6).

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<sup>4</sup> Personal communication with Dr Errol Thom, Scientist, Dairy NZ, Hamilton.

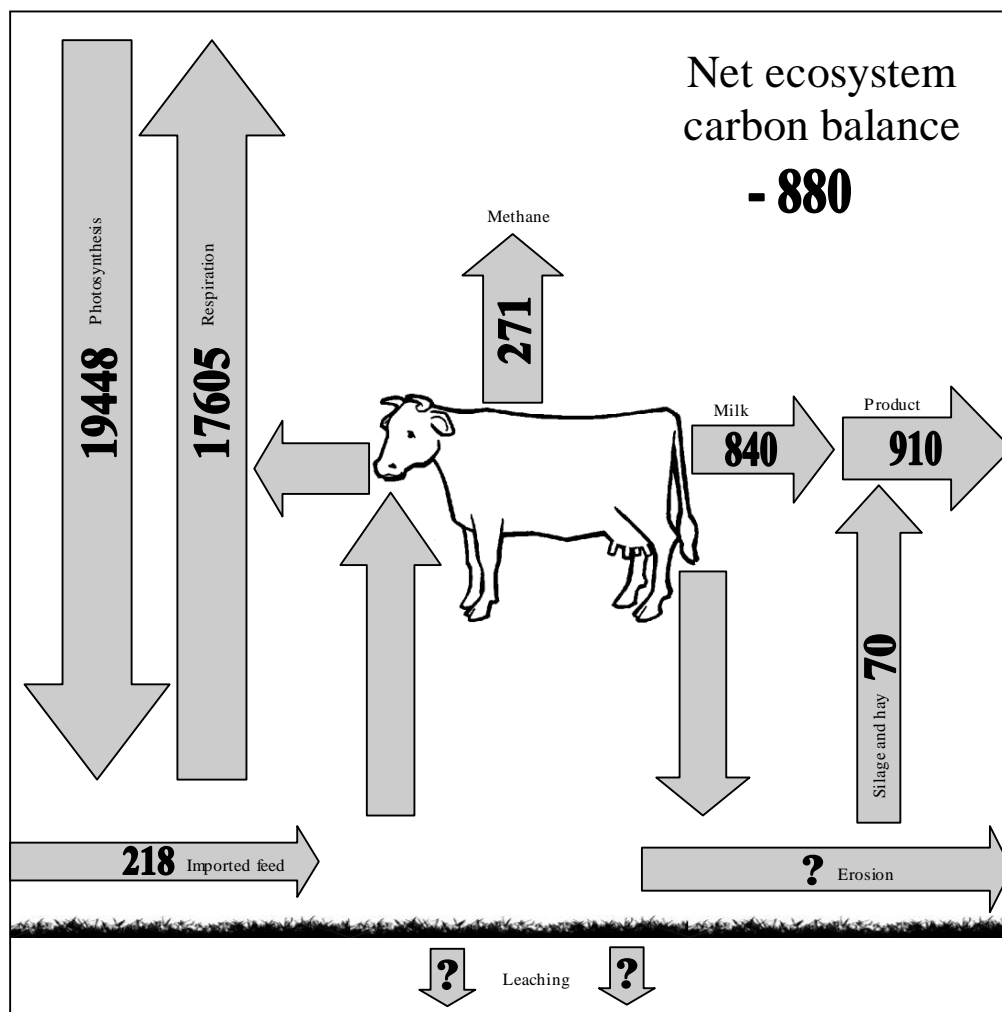


Fig. 3.6. Summary of inputs and outputs of C from the soil-pasture system at Scott Farm from 15 December 2007 to 14 December 2008. All values are in  $\text{kg C ha}^{-1} \text{y}^{-1}$ . The net ecosystem carbon balance of  $-880$  shows the soil-pasture system at Scott Farm was a net C sink.

### 3.3.3 Cultivation trial results

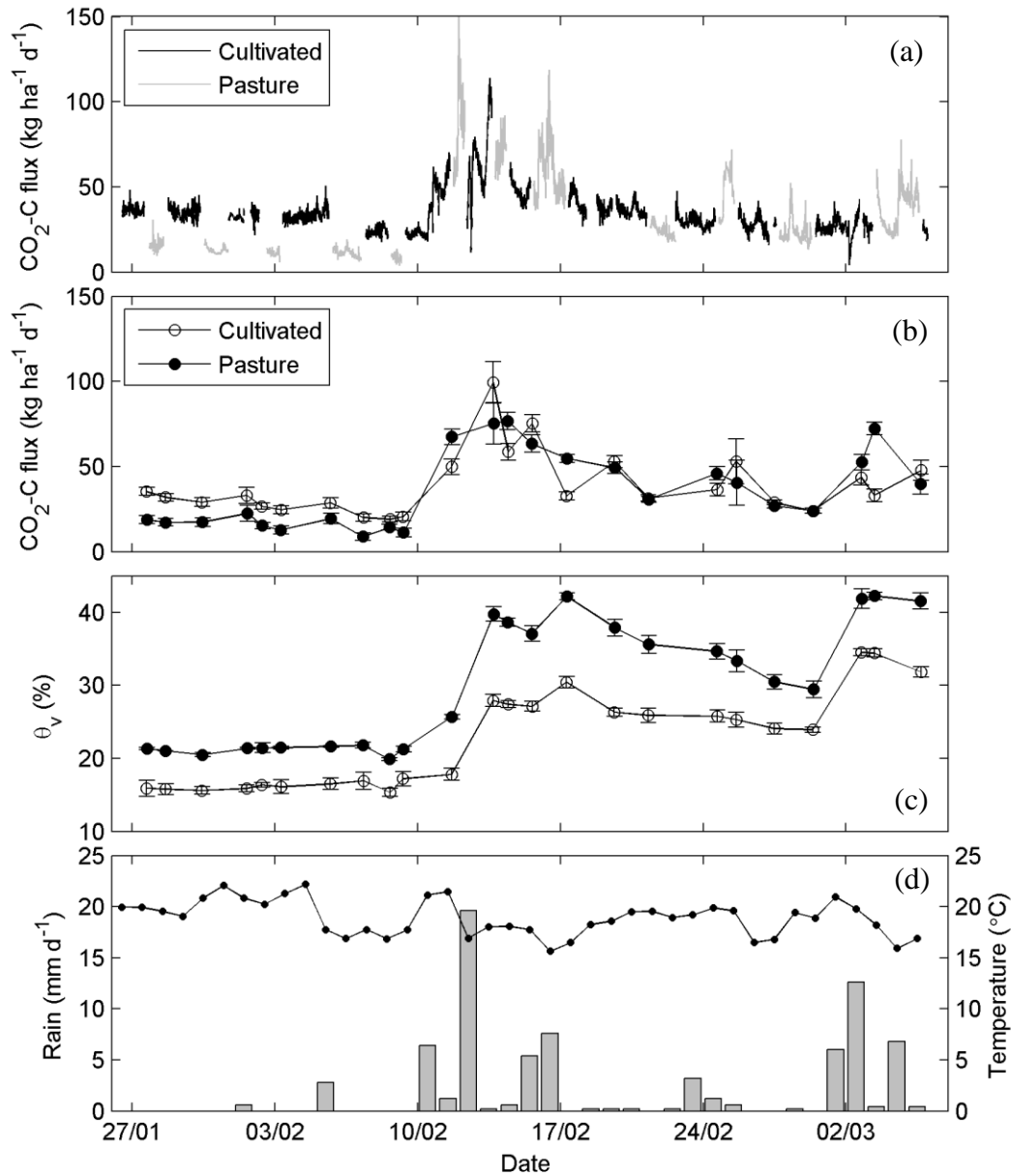
Cumulative  $\text{CO}_2$  emissions during the 39 day cultivation study were  $1,496 \text{ kg C ha}^{-1}$  from the cultivated paddocks ( $38.4 \text{ kg C ha}^{-1} \text{day}^{-1}$ ), and  $1,446 \text{ kg C ha}^{-1}$  from adjacent pasture paddocks ( $37.1 \text{ kg C ha}^{-1} \text{day}^{-1}$ ) (Table 3.1). These cumulative fluxes were not significantly different. Prior to rain on 10 February,  $\text{CO}_2$  emissions were significantly higher from the cultivated paddocks than pasture paddocks (Fig. 3.7a, b). Following rain on 10 February, there was a large increase in  $\text{CO}_2$  emissions from both cultivated and pasture paddocks, and subsequently there was no difference in emissions between treatments. This peak in emissions coincided with the peak in NEE (and calculated TER) measured with the EC system at the same time (Fig. 3.5a, b). Fluxes measured by the automatic chamber



showed the same trends as survey chamber measurements and were of the same magnitude (Fig. 3.7, Table 3.1). Automatic chamber measurements from the cultivated paddock showed that heavy rain events on 12 February and 2 March caused large decreases in CO<sub>2</sub> emissions for a short period after the rain event (Fig. 3.7a) (this will be discussed further in chapter 4). Soil moisture content was always higher in pasture paddocks than cultivated paddocks (Fig. 3.7c).

**Table 3.1:** Average daily respiration and total C respired during the 39 day cultivation study. Errors are 1 standard error (n=3).

Treatment	Chamber	Average respiration (kg C ha <sup>-1</sup> day <sup>-1</sup> )	Total C lost during study (kg ha <sup>-1</sup> )
Cultivated	Survey	38.4±2.8	1496±107
Pasture	Survey	37.1±1.0	1446±40
Cultivated	Automatic	34.6	1350
Pasture	Automatic	32.3	1260



**Fig. 3.7.** Soil respiration measured from cultivated and pasture paddocks using (a) an automatic chamber and (b) a survey chamber, along with (c) volumetric soil moisture content ( $\theta_v$ ) and (d) mean daily temperature (line) and total daily rainfall (bars). Error bars indicate  $\pm 1$  standard error ( $n=3$ ).

### 3.3.4 Pugging trial results

Cumulative  $\text{CO}_2$  emissions during the first 41 days following the pugging event, were  $1,161 \text{ kg C ha}^{-1}$  from the pugged plots ( $28.3 \text{ kg ha}^{-1} \text{ day}^{-1}$ ), which was significantly less than emissions from the control plots  $1,381 \text{ kg C ha}^{-1}$  ( $33.7 \text{ kg ha}^{-1} \text{ day}^{-1}$ ) (Fig. 3.8, Table 3.2). Average daily rates of  $\text{CO}_2$  emissions were similar to those from the cultivation trial (Table 3.1). Volumetric soil moisture content was always higher in the pugged plots (Fig. 3.8c), most likely because

infiltration was slower in the pugged plots than control plots (water often ponded on the surface of pugged plots but not control plots). A plot of cumulative 30 minute rainfall and CO<sub>2</sub> fluxes measured by the automatic chamber showed that rainfall events often coincided with a sharp decrease in CO<sub>2</sub> emissions (from both the collar in the pugged plot and the control plot). However, on some occasions rainfall events coincided with very large peaks in CO<sub>2</sub> emissions (e.g. the four largest peaks in Fig. 3.8a). These events will be further explored and discussed in chapter 4.

Automatic chamber measurements from the collar in the control plot were of similar magnitude to average survey chamber measurements from all control plots. However, automatic chamber measurements from the collar in the pugged plot were much lower than the average of survey chamber measurements from all the pugged plots (Fig. 3.8a). Automatic chamber measurements from the pugged plot were, however, well within the range of measurements from individual survey collars in the pugged plots.

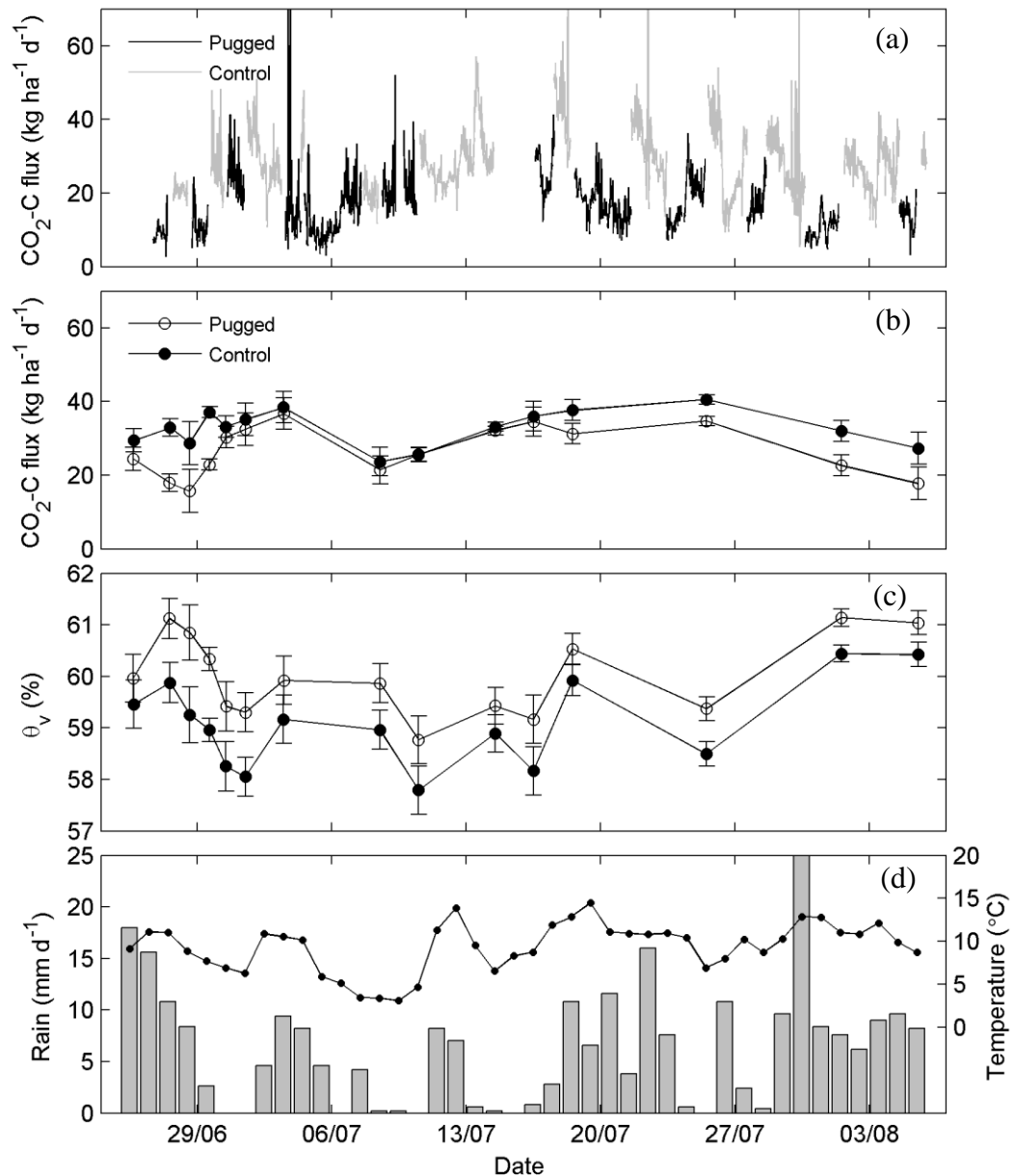
Prior to pugging, bare ground was the same in all plots (20%). Nine days after pugging, bare ground was significantly higher in the pugged plots (55%) than the control plots (23.8%). By the end of the study, bare ground had declined to 16.8% and 4.8% in the pugged and control plots respectively and were still significantly different. At the end of the study, total above ground biomass was significantly higher in the control plots (1,413 kg DM ha<sup>-1</sup>) than in the pugged plots (855 kg DM ha<sup>-1</sup>). Total pasture accumulation from individual collars over the duration of the study averaged 350 kg DM ha<sup>-1</sup> in the pugged plots and 437 kg DM ha<sup>-1</sup> in control plots, but this difference was not significant. However, excluding data from one collar in a pugged plot, which accumulated over twice as much biomass as any other collar, (due to being around the only surviving plant in a large area so it received lots of light) reduced average pasture accumulation in the pugged plots to only 264 kg DM ha<sup>-1</sup>. When this collar was excluded the difference in pasture accumulation between the two treatments was marginally significant (p=0.055). There was no significant difference in root biomass (0 – 75 mm depth) at the end of the study (1,622 and 1,855 kg DM ha<sup>-1</sup> in the pugged and control plots respectively). There was no significant difference in dry bulk density between the

two treatments measured on the 4 July (11 days after pugging) and at the end of the study (5 August). Volumetric soil moisture contents calculated from the bulk density cores collected on 4 July and 5 August were significantly higher in the pugged than control plots (consistent with measurements made with the Hydrosense probe, Fig. 3.8c).

There was a significant positive correlation between total pasture accumulation per collar and average CO<sub>2</sub> emissions per collar, with  $r^2$  values of 0.7 ( $p < 0.001$ ) and 0.52 ( $p < 0.001$ ) for collars in the control and pugged plots respectively (see Fig. C1 in Appendix C). A significant relationship was also found between total root biomass (in each collar) at the end of the study and cumulative CO<sub>2</sub> emissions with  $r^2$  values of 0.83 ( $p < 0.0011$ ) and 0.33 ( $p = 0.025$ ) for collars in the control and pugged plots respectively (see Fig. C2 in Appendix C).

**Table 3.2:** Average daily respiration and total C respired during the 41 day pugging study. Errors are 1 standard error (n=4).

Treatment	Chamber	Average respiration (kg C ha <sup>-1</sup> day <sup>-1</sup> )	Total C lost during study (kg ha <sup>-1</sup> )
Pugged	Survey	28.3±1.5	1161±63
Control	Survey	33.7±1.3	1381±54
Pugged	Automatic	16.1	659
Control	Automatic	28.7	1177



**Fig. 3.8.** Respiration measured from pugged and control plots using (a) an automatic chamber and (b) a survey chamber, along with (c) volumetric soil moisture content ( $\theta_v$ ) and (d) mean daily temperature (line) and total daily rainfall (bars). Error bars indicate  $\pm 1$  standard error ( $n=4$ ).

## 3.4 Discussion

### 3.4.1 Annual farm scale C budget

The annual NEE of  $-1,843 \text{ kg C ha}^{-1}$  indicates that the soil pasture system at Scott Farm was a net sink of atmospheric  $\text{CO}_2$ . This NEE was similar to the average NEE of  $-1,503 \text{ kg C ha}^{-1} \text{ y}^{-1}$  for 20 European grasslands (Gilmanov *et al.*, 2007). These European grasslands ranged from semi-natural grasslands in Mediterranean climates to intensively managed Atlantic grasslands. Greatest  $\text{CO}_2\text{-C}$  sink activity

occurred in Atlantic grasslands managed for hay, while grazed grasslands were either moderate CO<sub>2</sub>-C sinks or CO<sub>2</sub>-C neutral. However, average NEE of an intensive dairy farm in Ireland was  $-2,470 \text{ kg C ha}^{-1} \text{ y}^{-1}$  for three years (Jaksic *et al.*, 2006; Byrne *et al.*, 2007). In a grassland managed for hay in North Carolina NEE was  $970 \text{ kg C ha}^{-1} \text{ y}^{-1}$  (i.e. a C source) (Novick *et al.*, 2004) and average NEE (over four years) on an intensively managed grassland (cut and grazed) in Pennsylvania was  $270 \text{ kg C ha}^{-1} \text{ y}^{-1}$  and  $-650 \text{ kg C ha}^{-1} \text{ y}^{-1}$  for grassed based and alfalfa based pastures respectively (Skinner, 2008). NEE on a Waikato dairy farm, on drained peat soil was  $45 \text{ kg C ha}^{-1}$  (Nieveen *et al.*, 2005). Source activity on drained peat would be expected because increased oxygen inputs increase peat decomposition.

Despite the severe drought during the current study, annual NEE was still negative. In contrast, a number of studies have reported positive annual NEE values in years with severe drought. For example Aires *et al.* (2008), found that a grassland in Portugal was a net C source (NEE,  $490 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ) during a year with severe drought compared to being a strong sink of C (NEE,  $-1,900 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ) during a 'normal' year. Nagy *et al.* (2007) reported NEE values of  $800 \text{ kg C ha}^{-1} \text{ y}^{-1}$  and  $-1,880 \text{ kg C ha}^{-1} \text{ y}^{-1}$  during drought and normal years respectively, in a study on a semi-arid grassland in Hungary. Drought had a similar effect on annual NEE in a native prairie in Canada (Flanagan *et al.*, 2002) and in an extensively grazed tussock grassland in New Zealand (Hunt *et al.*, 2004). The warm temperate climate in the Waikato allows pastures to grow all year and therefore the effect of drought may be less than in cooler environments and annual grasslands where drought during the growing season has a large impact on annual NEE (Flanagan *et al.*, 2002; Hunt *et al.*, 2004; Xu and Baldocchi, 2004).

The calculated annual GPP of  $-19,448 \text{ kg C ha}^{-1} \text{ y}^{-1}$  and TER of  $17,605 \text{ kg C ha}^{-1} \text{ y}^{-1}$  were higher than most values reported from EC studies at other grassland sites. Gilmanov *et al.* (2007) found that GPP at 20 European grassland sites ranged from  $-4,636 \text{ kg C ha}^{-1} \text{ y}^{-1}$  "in dry semi-natural pastures to  $-18,816 \text{ kg C ha}^{-1} \text{ y}^{-1}$  in intensively managed Atlantic grasslands" with the average over all 20 sites being  $-12,612 \text{ kg C ha}^{-1} \text{ y}^{-1}$ . At the same 20 sites, TER ranged from 4,934 to 16,219  $\text{kg C ha}^{-1} \text{ y}^{-1}$  and averaged  $11,109 \text{ kg C ha}^{-1} \text{ y}^{-1}$  (Gilmanov *et al.*, 2007). In a warm

temperate grassland in North Carolina, GPP was  $-12,020 \text{ kg C ha}^{-1} \text{ y}^{-1}$  and TER  $12,990 \text{ kg C ha}^{-1} \text{ y}^{-1}$ . On an intensive dairy farm in Ireland, GPP and TER averaged over two years were  $-16,955$  and  $14,700 \text{ kg C ha}^{-1} \text{ y}^{-1}$  respectively (Jaksic *et al.*, 2006). GPP and TER at Scott Farm were also higher than on a nearby dairy farm (in the Waikato) on peat soil, where Nieveen *et al.* (2005) calculated a GPP of  $-13,486 \text{ kg C ha}^{-1} \text{ y}^{-1}$  and TER of  $13,531 \text{ kg C ha}^{-1} \text{ y}^{-1}$ . It would have been expected that GPP would have been similar to the study Nieveen *et al.* (2005) given that pastures and milk production and climate were similar.

GPP at Scott Farm would have likely been even higher in a year without drought. Cage cut data (Fig. 3.4) shows that above ground pasture production during the study period was lower than the two previous years and Aires *et al.* (2008) found that GPP and TER were about halved in a year with severe drought compared to a normal year. High GPP at Scott Farm was not surprising given the relatively warm temperate climate in the Waikato that favours year round pasture growth, in comparison to most European grasslands, where growth virtually ceases during winter.

Many eddy covariance studies do not take into account C fluxes other than  $\text{CO}_2$ -C exchange, to calculate the net C change of a pasture system. The soil-pasture system at Scott Farm was a C sink of  $-880 \text{ kg C ha}^{-1} \text{ y}^{-1}$  when all the other key inputs and outputs of C were accounted for. Quantifying C fluxes other than  $\text{CO}_2$ -C exchange measured by the EC system was important, because C export in milk ( $835 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ) combined with losses in methane ( $271 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ) were about 62% of the measured NEE ( $-1,843 \text{ kg C ha}^{-1} \text{ y}^{-1}$ ). Soussana *et al.* (2007) calculated total C budgets which included outputs of C in methane and harvested biomass and inputs of C in manure, at eight grassland sites in Europe (two years of data). They found that all sites except one were net C sinks with the average sink of  $-1,040 \text{ kg C ha}^{-1} \text{ y}^{-1}$  being significantly different from zero. However, they did not include C export in product and the current study and the study of Nieveen *et al.* (2005) have demonstrated that C loss in product can have a large influence on the annual C budget. Furthermore, Skinner (2008) found that including C exported in silage and manure inputs, meant that an alfalfa based

pasture in Pennsylvania was a source of  $480 \text{ kg C ha}^{-1} \text{ y}^{-1}$  rather than a sink of  $-650 \text{ kg C ha}^{-1} \text{ y}^{-1}$  when only considering NEE.

A major unknown for developing C balances for pasture systems is that C export via leaching is seldom included, predominantly because there have been few measurements of C leaching and losses can vary considerably depending on soil type. For example, Don and Schulze (2008) measured DOC leaching of  $8 \text{ kg C ha}^{-1} \text{ y}^{-1}$  in a clay rich Vertisol and  $55 \text{ kg C ha}^{-1} \text{ y}^{-1}$  in an Arenosol in an ungrazed grassland in Germany. In a four year lysimeter study in New Zealand (with four soils), Sparling *et al.* (2006) found that C leaching losses ranged from  $36.5 \text{ kg C ha}^{-1}$  in an allophanic soil to  $241.8 \text{ kg C ha}^{-1} \text{ y}^{-1}$  from a recent soil. Ghani *et al.* (2008) carried out an indoor incubation lysimeter study (for six months) on six different New Zealand soils collected from under intensive dairying. They calculated that DOC leaching would have been  $1,834$  and  $1,217 \text{ kg C ha}^{-1} \text{ y}^{-1}$  from two gley soils and between  $332$  and  $861 \text{ kg C ha}^{-1} \text{ y}^{-1}$  from four allophanic soils. Extrapolating C leaching losses measured in barrel lysimeters and glasshouse trials to field scale is questionable, but clearly more information on the magnitude of C leaching in the field is needed to finalize C budgets for pasture systems.

#### **3.4.1.1 Errors associated with the annual C budget**

During short time periods, random sampling errors associated with the EC technique can be relatively large (Richardson *et al.*, 2006) and further random error can be introduced in the gap filling process (Falge *et al.*, 2001; Moffat *et al.*, 2007). However, when measurements are averaged over an annual period these random errors are reduced to relatively small values (Baldocchi, 2003). Two systematic errors can also affect EC measurements. The first is that during calm conditions at night,  $\text{CO}_2$  emissions are usually underestimated (Baldocchi, 2003). Filtering out NEE measurements during these periods and subsequent gap filling as described in Section 3.2.2.4 is currently regarded as the best way to eliminate this error (Aubinet, 2008). The second systematic error is that energy balance closure is usually underestimated by EC measurements, which suggests that  $\text{CO}_2$  fluxes may also be underestimated (Baldocchi, 2003). Annual energy balance closure at Scott Farm was underestimated by 19% in a parallel study (Kuske,



2009), which is within the typical range for eddy covariance studies (Wilson *et al.*, 2000; Baldocchi, 2003). However, CO<sub>2</sub> fluxes at Scott Farm were not corrected for lack of energy balance closure, in accordance with most other recent EC studies (Nieveen *et al.*, 2005; Ammann *et al.*, 2007; Aires *et al.*, 2008). The accuracy of annual sums of NEE measured using the EC technique typically range from 300 to 1,000 kg ha<sup>-1</sup> y<sup>-1</sup> (Baldocchi, 2008). Baldocchi (2003) explains that at nearly ideal sites, error bounds are generally less than  $\pm 500$  kg C ha<sup>-1</sup> y<sup>-1</sup>. Site conditions at Scott Farm were good for EC measurements, with flat topography and unimpeded fetch of >200 m in every direction. Therefore, following the approach of Nieveen *et al.* (2005), an estimated confidence range for annual NEE of  $\pm 500$  kg C ha<sup>-1</sup> y<sup>-1</sup> seems appropriate for this study.

Errors associated with C export in milk will likely be small, because milk quantity is measured daily from each cow on Scott Farm and milk composition tests are conducted weekly. Errors associated with C losses in methane are not known but are likely to be relatively small in terms of C budget accounting. The relatively small amount of C exported in harvested pasture and imported in palm kernel extract suggests errors associated with these fluxes will also be small. Therefore the final paddock scale C balance would have been  $-880 \pm 500$  kg C ha<sup>-1</sup> y<sup>-1</sup> (not accounting for C leaching).

### 3.4.2 Seasonal trends

A full analysis of factors controlling CO<sub>2</sub> exchange is given in Chapter 4, here a preliminary analysis and discussion of seasonal trends is provided.

On average, the soil-pasture system was a net C source from 15 December to the end of April, after which it became a net C sink (Fig. 3.5d). During the period from 30 December 2007 to 14 April 2008 volumetric soil moisture content was below 40% (usually well below, Fig. 3.3c). For soils very similar to the Matangi silt loam (e.g. Bruntwood silt loam and Te Kowhai silt loam, located adjacent to the Matangi soil on Scott Farm), Singleton, (1991) found that 40 – 45% was the lower limit of readily available water, below which plants begin to become moisture stressed. Therefore the C source activity during the drought period (soil

moisture <40%) was probably caused because low soil moisture content reduced photosynthesis proportionally more than respiration (Nagy *et al.*, 2007).

The seasonal pattern of NEE at Scott Farm was similar to the study of Nieveen *et al.* (2005) for a pasture on peat soil in the Waikato. During the late summer-autumn period of their study, soil moisture was also low and the soil-pasture system was a C source, while in spring the soil-pasture system was a strong C sink. In a tussock grassland in southern New Zealand, Hunt *et al.* (2004) found that a summer drought caused a large loss of C, while the following year, during the same time period, when soil moisture was 'normal' the tussock grassland was either a C sink or C neutral.

Harvesting and grazing have been shown to cause grasslands to quickly change from being a C sink to a C source (Novick *et al.*, 2004; Nieveen *et al.*, 2005; Jaksic *et al.*, 2006). The large scale harvesting of pasture from within the flux footprint during January may have contributed to the soil-pasture system becoming C source in late January – early February. However, the very low soil moisture contents during this period (Fig. 3.3) meant that pastures had practically stopped growing and therefore the C source activity was attributed mostly to the dry conditions, rather than pasture harvest.

### 3.4.3 *CO<sub>2</sub> emissions following cultivation*

There was no significant difference in cumulative CO<sub>2</sub> emissions from the cultivated and pasture paddocks during the 39 day study (Table 3.1). Average CO<sub>2</sub>-C emissions from the cultivated (38.4 kg ha<sup>-1</sup> day<sup>-1</sup>) and pasture paddocks (37.1 kg ha<sup>-1</sup> day<sup>-1</sup>) were of similar magnitude to other studies of pasture and/or cultivated soils (Aslam *et al.*, 2000; Grandy and Robertson, 2006; Quincke *et al.*, 2007; Chatskikh *et al.*, 2008).

While there was no difference in cumulative CO<sub>2</sub> emissions for the whole study, CO<sub>2</sub> emissions were significantly higher from cultivated paddocks prior to rain on 10 February. Grandy and Robertson (2006) found that CO<sub>2</sub>-C emissions from a field that had never been tilled in Michigan (USA) averaged 32.2 kg ha<sup>-1</sup> day<sup>-1</sup> compared to 48.6 kg ha<sup>-1</sup> day<sup>-1</sup> from the same field cultivated by mouldboard

ploughing. They attributed the higher emissions from the cultivated plots to higher soil temperatures and the destruction of aggregates which increased the amount of labile C available for microbial consumption. An increase in labile C could have contributed to higher emissions from the cultivated paddocks at Scott Farm, prior to rain, but this does not explain why emissions from the two treatments were similar following rain. Another explanation for the higher emissions from cultivated paddocks prior to rain, could be that the inversion of the soil following ploughing placed labile C from near the surface in contact with slightly moister soil at 200 mm depth, which may have stimulated microbial activity.

Most other studies have reported significantly higher CO<sub>2</sub> emissions from pasture than cultivated paddocks. For example, Aslam *et al.* (2000) measured CO<sub>2</sub> emissions for a year (about once per month) from permanent pasture in New Zealand and paddocks double cropped with maize and oats (using mouldboard ploughing). CO<sub>2</sub>-C emissions from Aslam's year-long study were usually significantly greater in the permanent pasture treatment (average 92 kg C ha<sup>-1</sup> day<sup>-1</sup>), than from the conventionally tilled treatment (average 63 kg C ha<sup>-1</sup> day<sup>-1</sup>). Similarly, Lohila *et al.* (2003) found that soil respiration was generally higher from soil under pasture than under barley in Finland. During a short 21-day study in the UK, Yamulki and Jarvis (2002) measured average CO<sub>2</sub> emissions of 76 kg C ha<sup>-1</sup> day<sup>-1</sup> from pasture compared to 43 kg C ha<sup>-1</sup> day<sup>-1</sup> from the same pasture that had been cultivated to 200 mm depth. Greater CO<sub>2</sub> emissions from the pasture soils in these studies was probably due to higher root biomass and therefore greater root respiration and respiration of microbes consuming root exudates (Lohila *et al.*, 2003). In the current study, the contribution of pasture plants to total soil respiration would have likely been limited because plants were moisture stressed during the drought.

The non-significant difference in cumulative CO<sub>2</sub>-C emissions between the cultivated and pasture paddocks does not mean that there was no difference in net C loss between the two treatments. C inputs via photosynthesis to the pasture paddocks (when cultivated paddocks remained bare) must also be considered. Cumulative TER at the EC site during the cultivation study was 1,262 kg C ha<sup>-1</sup>. This was similar to values calculated from measurements made in the pasture

paddocks of the cultivation trial with the survey and automatic chambers (1,446 and 1,260 kg C ha<sup>-1</sup> respectively (Table 3.1)). Cumulative GPP measured with the EC system during the 39 day cultivation trial was -572 kg C ha<sup>-1</sup>. Assuming GPP in the three pasture paddocks in the cultivation study was the same as around the EC tower (a fair assumption given they were on the same soil type and were part of the same farmlet trials and under the same management) the net C loss from the pasture paddocks would have been about 874 kg ha<sup>-1</sup> (i.e. 1,446 kg C ha<sup>-1</sup> respired minus 572 kg C ha<sup>-1</sup> fixed via photosynthesis). There was no photosynthetic input to the cultivated paddocks during the trial and therefore the cumulative CO<sub>2</sub>-C emissions of 1,496 kg ha<sup>-1</sup> was a net loss from the system. Therefore, when photosynthetic inputs to the pasture paddocks are included in calculations the cultivated paddocks lost about 622 kg C ha<sup>-1</sup> (16 kg C ha<sup>-1</sup> day<sup>-1</sup>) more than the adjacent pasture paddocks during the 39 day study.

The calculated net C loss of approximately 622 kg C ha<sup>-1</sup> following cultivation will likely be an underestimate of the total C loss because C loss between spraying and ploughing (~10 days) and between ploughing and power harrowing (~7 days) was not quantified. The dry conditions also meant that that growth of the new pastures was very limited prior to May, leaving a prolonged period when there was little input of C to the soil via photosynthesis. Knowledge of the recovery rate of soil C once new pastures have been established is also required if the long term effect of cultivation for pasture renewal on soil C is to be determined. Schipper and Sparling (submitted-a) found that for 10 New Zealand soils, C accumulated at an average rate of 1,070 kg C ha<sup>-1</sup> y<sup>-1</sup> during the first five years following conversion of scrub or old pastures to high producing pasture. Shepherd *et al.* (2001) found a similar accumulation rate of 1,080 kg C ha<sup>-1</sup> y<sup>-1</sup> for the Manawatu silt loam, but only 50 kg C ha<sup>-1</sup> y<sup>-1</sup> for the Kairanga silt loam during the first 10 years after conversion of cropping land to permanent pasture.

These accumulation rates, combined with the relatively small C loss of 622 kg C ha<sup>-1</sup> caused by the cultivation event and the fact that cultivation of pasture for fodder crops or pasture renewal occurs only about every 10 – 15 years in New Zealand, suggests that such cultivation events are unlikely to cause a large decline in soil C. However, the current study was conducted during severe moisture

limitation and further information is needed on the magnitude of CO<sub>2</sub>-C losses after cultivation under wetter conditions, when the rate of soil C decomposition is likely to be greater.

The minimum C loss caused by the cultivation event of 622 kg C ha<sup>-1</sup> would have reduced the farm scale C sink from -880 kg C ha<sup>-1</sup> to -252 kg C ha<sup>-1</sup>, if all paddocks surrounding the EC system were cultivated. However, this is assuming that once new pastures became established the C balance was the same as the established pastures around the EC tower. Analysis of cage cut data from around the EC tower revealed that the average pasture growth rate (above ground) of the few pasture paddocks sown in April 2008, was similar to the old pastures (53 and 45 kg DM ha<sup>-1</sup> day<sup>-1</sup> respectively, measurements of the new pastures only commenced once pastures were fully established in July). This partially supports the assumption that the C balance of new pastures is similar to old pastures, although allocation of C below ground is often greater in young plants (Kuznyakov, 2001).

#### **3.4.4 CO<sub>2</sub> emissions following pugging**

Higher CO<sub>2</sub>-C emissions from the control than pugged plots was in agreement with Hynst *et al.* (2007) who measured CO<sub>2</sub>-C emissions at three points along a gradient of cattle impact in a cattle overwintering area in the Czech Republic. They found that CO<sub>2</sub>-C emissions were 20 kg C ha<sup>-1</sup> day<sup>-1</sup> (averaged over two years) in the most highly impacted area near the wintering barn, 26.2 kg C ha<sup>-1</sup> day<sup>-1</sup> in a moderately impacted area and 42 kg C ha<sup>-1</sup> day<sup>-1</sup> from an area where the soil and pasture were largely un-impacted. Hynst *et al.* (2007) suggested that reduced aeration due to damaged soil structure was one possible reason for the decreased emissions from the highly impacted areas. It is likely that in the current study, similar mechanisms were responsible for decreased CO<sub>2</sub> emissions from pugged plots. There was no difference in soil bulk density between the pugged and control plots, indicating that total porosity was the same, however, pore size distribution had likely been altered with fewer macropores (Nguyen *et al.*, 1998; Zegwaard, 2005). This suggestion is consistent with the observation of water ponding on the surface of pugged plots after rain but not the control plots and the

consistently higher soil moisture in the pugged plots (Fig. 3.8). The wetter soil conditions in the pugged plots would have potentially limited oxygen availability to microbes and soil fauna which may have reduced soil respiration (Ball *et al.*, 1999; Davidson *et al.*, 2000). The observation of increased denitrification (Menneer *et al.*, 2005a) and N<sub>2</sub>O production (Thomas *et al.*, 2008) from pugged soils in other studies supports the suggestion that oxygen availability may have contributed to the lower respiration rates from the pugged plots.

Another possible cause of the lower CO<sub>2</sub> emissions from the pugged plots was that pasture production was lower in the pugged than control plots. Cumulative CO<sub>2</sub> emissions were positively correlated with cumulative pasture production. Therefore it was not surprising that the pugged plots with lower pasture production had lower CO<sub>2</sub> emissions. Hynst *et al.* (2007) similarly noted that plant biomass decreased as treading increased and suggested that decreased root respiration may have contributed to the lower CO<sub>2</sub> emissions they observed from the impacted areas. Clipping and shading experiments (to prevent photosynthate being translocated to roots) in grasslands have shown that rhizosphere respiration can contribute 20 – 44% of total soil respiration (Craine *et al.*, 1999; Wan and Luo, 2003; Byrne and Kiely, 2006).

These results indicate cattle pugging is unlikely to stimulate large losses of soil carbon via CO<sub>2</sub>. However, a pugging event followed by more ‘normal’ rainfall may have a different effect and the longer term effects of pugging on soil C are unknown. For example, Zegwaard (2005) found that during the first 22 weeks following severe pugging, cumulative herbage accumulation was reduced by 3,940 kg DM ha<sup>-1</sup> compared to unpugged control plots. Whether such a reduction in herbage production would contribute to a decrease in soil C is not known.

### 3.5 Conclusions

This study is one of few studies of carbon exchange in intensively grazed temperate pasture systems. Annual NEE measured with the eddy covariance system was -1,843 kg C ha<sup>-1</sup> and when all the key components of the C budget were included the soil-pasture system was net sink of -880±500 kg C ha<sup>-1</sup>. This C

sequestration occurred despite a severe drought during the first part of the study, which was in contrast to other studies of grasslands during drought (Flanagan *et al.*, 2002; Hunt *et al.*, 2004; Nagy *et al.*, 2007; Aires *et al.*, 2008). This difference may have been caused because the warm temperate climate in the Waikato allows pastures to grow all year and therefore C losses during the drought were recovered during the winter and spring.

The single cultivation event under dry conditions did not increase cumulative CO<sub>2</sub>-C emissions compared to adjacent pasture paddocks. However, when C inputs to pasture paddocks via photosynthesis were included in calculations, net C loss from the cultivated paddocks was about 622 kg ha<sup>-1</sup> more than the pasture paddocks. CO<sub>2</sub> emissions were lower from the pugged plots compared to the control plots which was probably caused by the combination of decreased microbial respiration due to wetter soil conditions and lowered root respiration, as a result of lower pasture production.

These results suggest that the large C losses observed by Schipper *et al.* (2007) from dairy farm soils were unlikely to have been caused by a one off severe summer/autumn drought, a single cultivation event for pasture renewal or a single soil pugging event by cattle in winter. However, the effect of drought, cultivation and pugging, may be different on other soil types or in other climatic regions of New Zealand. The cultivation study was also carried out when soils were very dry and further information is needed on the effect of cultivation when soils are moister.





## 4. ENVIRONMENTAL CONTROLS OF CO<sub>2</sub> EXCHANGE IN AN INTENSIVELY GRAZED PASTURE

### 4.1 Introduction

Carbon inputs via photosynthesis and outputs via respiration are generally the largest fluxes of C to and from pastoral ecosystems (Soussana *et al.*, 2007). Therefore understanding the factors that control the magnitude of these two processes is important because it may allow the development of farm systems that maximize photosynthetic uptake of C and minimize respiration and therefore improve C sequestration in soil. Understanding factors controlling photosynthesis may also provide insight into approaches that maximize pasture production (e.g. identifying optimum soil moisture content).

The carbon balance of an ecosystem is largely determined by the balance between inputs of CO<sub>2</sub>-C from gross primary production (GPP) and losses of CO<sub>2</sub>-C from total ecosystem respiration (TER). In terms of the C balance of soil-pasture systems, the absolute values of GPP and TER are less important than the balance between these two fluxes, which is termed net ecosystem exchange (NEE). Eddy covariance systems provide a direct measure of NEE, from which GPP and TER can be partitioned using modelling approaches (Reichstein *et al.*, 2005; Gilmanov *et al.*, 2007). In this chapter, gains of CO<sub>2</sub> by the land surface will be denoted by negative NEE values and losses to the atmosphere as positive NEE values. At night, NEE is a direct measure of TER, while during the day NEE is the balance between GPP and TER. Therefore, during the day when NEE is negative, GPP must be greater than TER (i.e. the land surface is a C sink) and if NEE is positive, TER must be greater than GPP (i.e. the land surface is a C source). Due to the short height of pasture vegetation, chamber systems can also be used to measure TER from pasture systems (excluding animal respiration).

Gross primary production (GPP) via photosynthesis is strongly affected by temperature and soil moisture, but light is an additional requirement for photosynthesis (Gilmanov *et al.*, 2007). Plant factors such as species and

vegetative state (e.g. pre and post-grazing) along with nutrient availability can also influence rates of photosynthesis. Vapour pressure deficit (VPD) is another factor that can affect photosynthesis. When soil moisture is limiting and VPD is high, plants tend to close their stomata to reduce water loss and hence rates of photosynthesis are reduced (Hunt *et al.*, 2002).

Total ecosystem respiration (TER) can be separated into two components: autotrophic respiration (AR) which is plant respiration, and heterotrophic respiration (HR) which is respiration of heterotrophic organisms (microbes and animals) (Chapin *et al.*, 2006). A range of environmental factors affect respiration rates, including temperature, moisture, substrate availability and quality, soil pH and oxygen availability (Davidson and Janssens, 2006; Luo and Zhou, 2006). TER is usually positively correlated with GPP, because with greater photosynthesis both AR and HR are enhanced (Craine *et al.*, 1999; Luo and Zhou, 2006; Aires *et al.*, 2008; Baldocchi, 2008).

Soil moisture content and temperature are generally the two key factors controlling respiration (Franzluebbers *et al.*, 2002; Yuste *et al.*, 2007), with light being an additional key control on photosynthesis. Separating the relative effect that each of these three factors has on respiration or photosynthesis is difficult, due to strong interactions between the three variables and the influence of multiple other factors (Luo and Zhou, 2006). For example, light and temperature are usually correlated (e.g. warmer temperatures during the day) and soil temperatures are generally warmer when soil moisture content is low (e.g. during drought). Respiration can be strongly affected by C inputs to roots via photosynthesis which in turn is affected by light, temperature and moisture. Tang *et al.* (2005) demonstrated that peak soil respiration under an oak tree lagged peak photosynthesis by 7 – 12 hours. There can also be lag effects of plants relative to microbes when environmental conditions change (Baldocchi, 2008). For example, rain after drought almost immediately stimulates microbial respiration, but plants often take longer to respond to replenished soil moisture (Hunt *et al.*, 2004).

The previous chapter examined the annual C budget and the effect that large scale perturbations that drought, cultivation and cattle pugging had on the C budget of a

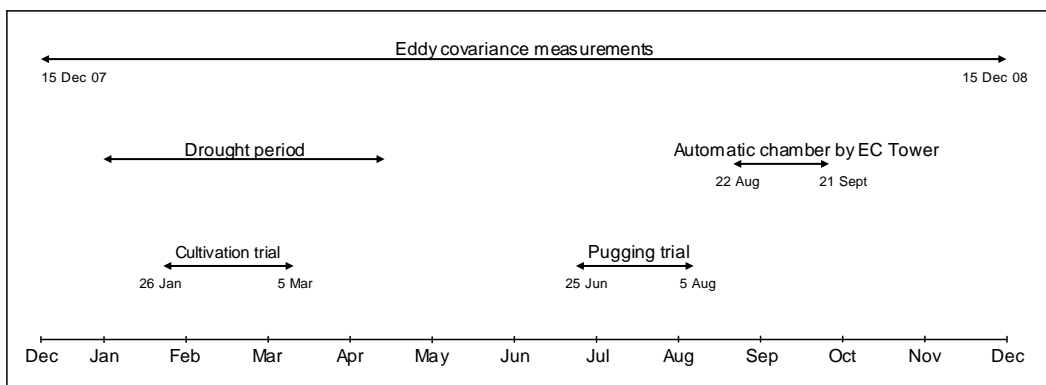
dairy farm soil-pasture system (Scott Farm). The objective of this chapter is to explore further the effect that temperature, moisture and light (photosynthetic photon flux density, PPFD) had on CO<sub>2</sub> exchange between the land surface and atmosphere at Scott Farm. Results will be presented from eddy covariance measurements and chamber measurements made in the cultivation and pugging experiments described in Chapter 3. It is recognized that there are controlling factors other than temperature, moisture and light, such as grazing and pasture harvest that will also have affected CO<sub>2</sub> exchange at Scott Farm. However, these effects were hard to disentangle due to the numerous small paddocks surrounding the EC tower, each grazed and/or harvested at different times and therefore the focus will be predominantly on temperature, moisture and light.

#### ***4.1.1 Chapter outline***

The chapter will begin with a brief review of methods used and the annual time series of eddy covariance (EC) data (more fully described in Chapter 3). The effects rainfall and soil moisture had on CO<sub>2</sub> exchange will then be investigated, with results from annual to daily time scales presented and discussed. The effect that temperature and PPFD had on CO<sub>2</sub> exchange will then be explored. Finally, pseudocolour graphs will be presented to explore the interactive relationships between NEE, and the controlling factors of temperature, moisture and PPFD. The focus will be on data from the eddy covariance (EC) system, with data from chamber measurements presented to compliment the EC data.

## **4.2 Methods**

This study was conducted at a research dairy farm (Scott Farm, owned and operated by DairyNZ) located 7 km northeast of Hamilton in the Waikato, North Island, New Zealand. Full site descriptions and methods are presented in Chapter 3. However, methods will be briefly summarized below along with a few additional details that are relevant to this chapter. A summary diagram showing when the different components of this research were carried out is presented in Fig. 4.1.



**Fig. 4.1.** Timeline showing when different components of this study were conducted. Also shown is the drought period.

### 4.2.1 Eddy covariance study

An eddy covariance system (EC) was established in an open pasture area of Scott Farm on 15 December 2007 (Fig. 3.1), and measurements were made continuously (a few minor gaps) until 14 December 2008 (further details are provided in Chapter 3. Section 3.2.2). Fluxes of CO<sub>2</sub> were calculated online (from 20 Hz signals) and stored as 30 minute averages. Standard corrections were made to these data and filtering applied to remove spurious data (e.g. during rain and low windspeed). Gaps were filled and NEE partitioned between Gross Primary Productivity (GPP) and Total Ecosystem Respiration (TER) using the online software described by Reichstein *et al.* (2005). A description of the gap filling and flux partitioning methods of Reichstein *et al.* (2005) is given in Appendix B. In addition to CO<sub>2</sub> exchange data, a range of other meteorological and soil variables were also measured at the EC site. These were also stored as 30 minute averages, except rain which was a 30 minute sum. Key variables that were used in the data analysis for this chapter were rain, air temperature, photosynthetic photon flux density (PPFD), soil temperature (50 mm) and volumetric soil moisture (average of 50 and 100 mm depths). The sensor for PPFD began drifting slightly after 24 February 2008 and therefore a regression between PPFD and global radiation (R<sub>g</sub>) from 15 December to 24 February was carried out ( $r^2 = 0.999$ ) and PPFD for the whole dataset was predicted from R<sub>g</sub> based on this relationship (Nieveen *et al.*, 2005). Between 22 August and 21 September, automatic chamber measurements were made next to the EC tower. Measurements were made in the same way and

using the same equipment used in the cultivation and pugging experiments (Chapter 3, Section 3.2).

#### **4.2.2 *Cultivation trial***

The cultivation trial was comprised of six paddocks (3 cultivated and 3 pasture controls) on the Matangi soil (but outside the EC flux footprint, see Fig. 3.1). Between 26 January and 5 March CO<sub>2</sub> emissions were measured about every second day from 10 collars (100 mm diameter) in each paddock with a survey chamber (60 collars total). CO<sub>2</sub> emissions were also measured every 10 minutes using an automatic chamber which was moved on alternate days between one collar (200 mm diameter) in a cultivated paddock and one collar in an adjacent pasture paddock. Soil temperature (50 mm depth), air temperature and volumetric soil moisture content (average 0 – 120 mm depth, for the survey chamber and at 50 mm depth adjacent to the automatic chamber) were measured at each collar each time CO<sub>2</sub> emissions were measured.

#### **4.2.3 *Pugging trial***

During winter, a pugging trial was carried out on the Matangi soil about 250 m from the EC tower (Fig. 3.1). The trial used eight small plots arranged in a randomized block design, with four blocks and two treatments (pugged and unpugged). Each plot was 7 m by 7 m, with a 2 m buffer strip around plot margins. Half of the plots were pugged once (24 June 2008) by 3 pregnant Friesian dairy cows per plot (~500 kg, 612 cows ha<sup>-1</sup>) over a 5 hour period. CO<sub>2</sub> emissions, temperature and soil moisture content were measured using the same instrumentation and methods described for the cultivation experiment. CO<sub>2</sub> emissions were measured every day (except day 2) for the first 7 days, then every 2 – 3 days for 2 weeks and then weekly for 3 weeks.

#### 4.2.4 Respiration and light response equations

The Lloyd and Taylor (1994) equation is an exponential equation commonly used to model soil respiration. This equation was fitted to NEE and chamber CO<sub>2</sub> flux data to see how a theoretically ‘correct’ model fitted the data. The equation is shown below.

$$TER = R_{10} e^{308.56 \left( \frac{1}{56.02} - \frac{1}{Ts - 46.02} \right)} \quad \text{equation 4.1}$$

where  $R_{10}$  is respiration at 10 °C and  $T_s$  is soil temperature (°C) at 50 mm.

The relationship between 30 minute averages of NEE and PPFD was assessed using the Michaelis-Menten rectangular hyperbola (Falge *et al.*, 2001; Aires *et al.*, 2008):

$$NEE = \frac{GPP_{max} PPFD}{K + PPFD} + TER \quad \text{equation 4.2}$$

where  $GPP_{max}$  is gross primary production at infinite light and  $K$  is the PPFD level which corresponds with half of  $GPP_{max}$ . Apparent quantum yield ( $\alpha$ ) is calculated as  $GPP_{max}/K$ .

### 4.3 Results and discussion

#### 4.3.1 Description of eddy covariance annual time series

Fig.4.2 shows an annual time series of daily average GPP, TER, NEE, soil moisture content and rain along with average monthly NEE and cumulative NEE (adapted from Fig. 3.4) This figure is presented here and explained again to provide context for this chapter.

Between 26 December 2007 and 9 February, both total ecosystem respiration (TER) and gross primary production (GPP) decreased rapidly as the soil dried out (Fig. 4.2a, c). By the end of January, GPP had practically ceased, TER had declined to low levels and NEE was positive indicating that the soil-pasture system had become a source of C (Fig. 4.2b). Rain between 10 and 17 February

caused a large increase in TER, but GPP remained low which meant the soil-pasture system was a large C source during this period (Fig. 4.2a, b). TER remained higher than GPP until May, meaning that the soil-pasture system was a net C source during the months of February, March and April (Fig. 4.2d). In contrast to April, during May the soil-pasture system was a large net sink of C, while in June the soil-pasture system was a small C sink. From July until the end of September, both TER and GPP gradually increased. During this period, the soil-pasture system was a net sink of C, with sink intensity increasing from June – September (Fig. 4.2d). TER remained relatively constant from the end of September to 14 December, while GPP fluctuated considerably, but in general, remained relatively high. Maximum monthly C sequestration occurred in November. The fluctuations in GPP after September were possibly caused by the frequent grazing and silage harvesting events during this period (Nieveen *et al.*, 2005). However, due to the many small paddocks within the flux footprint, coupled with changes in wind direction it was not possible to confirm that this was indeed the cause of the fluctuations.

The large decline in TER and GPP observed during the most severe drought period is consistent with other grassland studies conducted during drought (Hunt *et al.*, 2004; Nagy *et al.*, 2007; Aires *et al.*, 2008) and also an analysis of data from 14 eddy flux sites (mainly forest) in Europe during the 2003 European heatwave (Reichstein *et al.*, 2007). The loss of C that commonly occurs during drought periods is usually because GPP declines more than TER (Nagy *et al.*, 2007), which is presumably due to microbes being able to survive (and remain active) at lower soil moisture contents than plants. Although only data for one season was available, the seasonal extremes of GPP and TER appeared to be less pronounced at Scott Farm than in studies conducted in cooler climates, where GPP and TER decline to very low levels in the winter (Allard *et al.*, 2007; Ammann *et al.*, 2007).

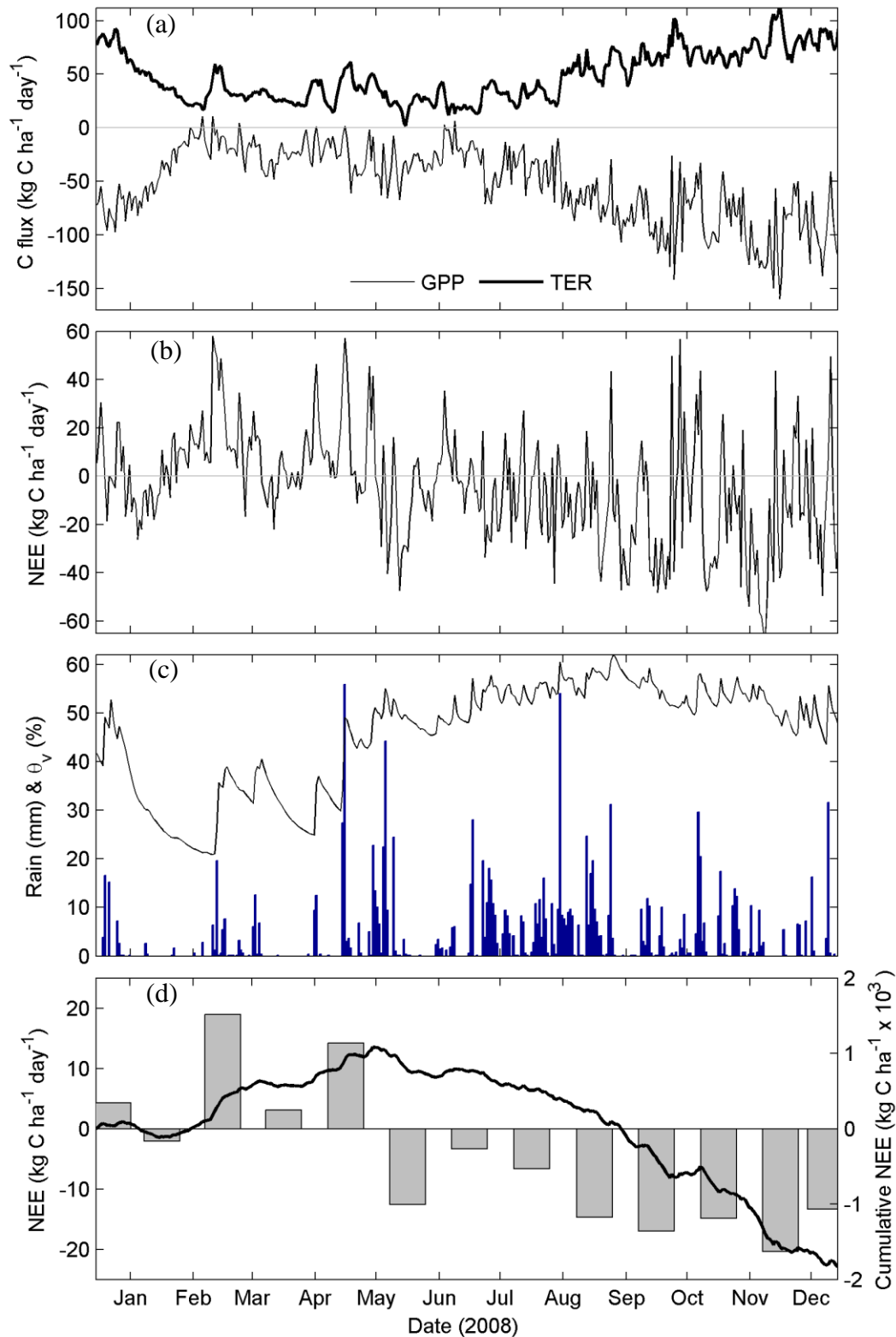


Fig. 4.2. Daily totals of (a) modelled gross primary production (GPP, plotted as negative values) and total ecosystem respiration (TER), (b) daily totals of gap filled net ecosystem exchange (NEE), (c) daily mean volumetric soil moisture content ( $\theta_v$ ) line, and sum of daily rain (bars), (d) mean monthly NEE (bars) and cumulative NEE (line), from 15 December 2007 to 14 December 2008.



### ***4.3.2 The effect of rain and soil moisture content on CO<sub>2</sub> exchange***

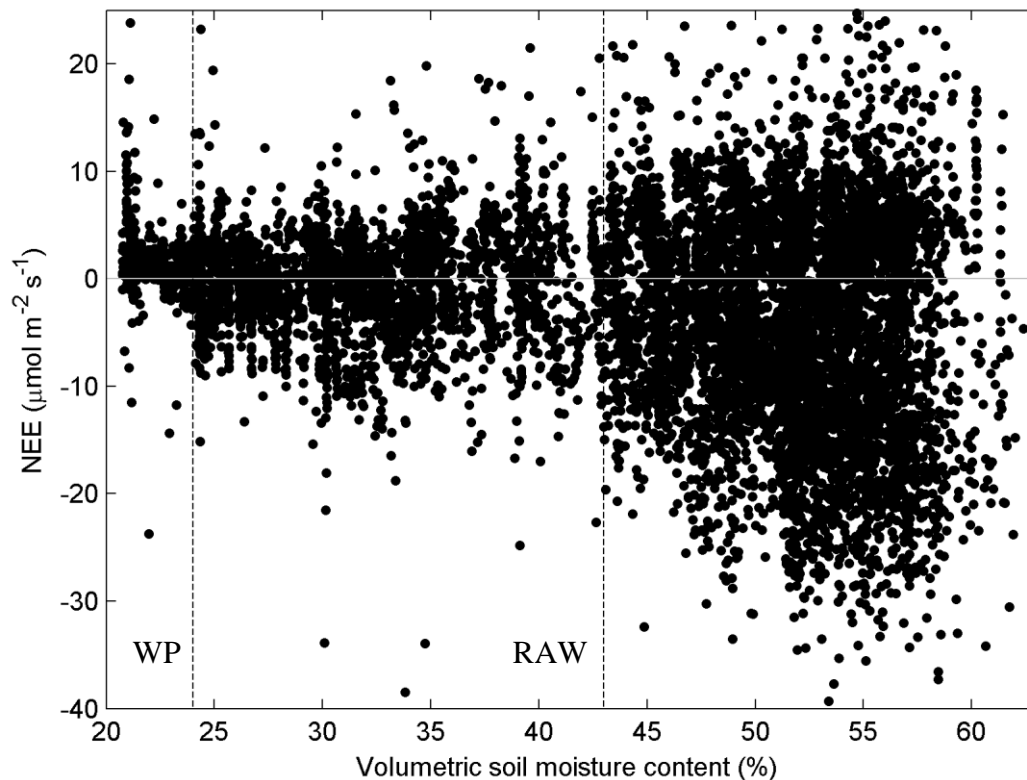
Volumetric soil moisture content (hereafter, soil moisture) can directly influence CO<sub>2</sub> exchange by affecting physiological processes in plants and microbes and also indirectly by influencing diffusion rates of oxygen and substrates (Davidson and Janssens, 2006; Luo and Zhou, 2006). Soil respiration is generally only restricted by soil moisture when soils are very wet or very dry (Fang and Moncrieff, 1999; Davidson *et al.*, 2000; Fang and Moncrieff, 2001; Luo and Zhou, 2006). Results from this study suggest that soil moisture content at Scott Farm reached both the lower and upper limits that affect respiration, while photosynthesis was also clearly limited by low soil moisture conditions.

#### ***4.3.2.1 Long term trends***

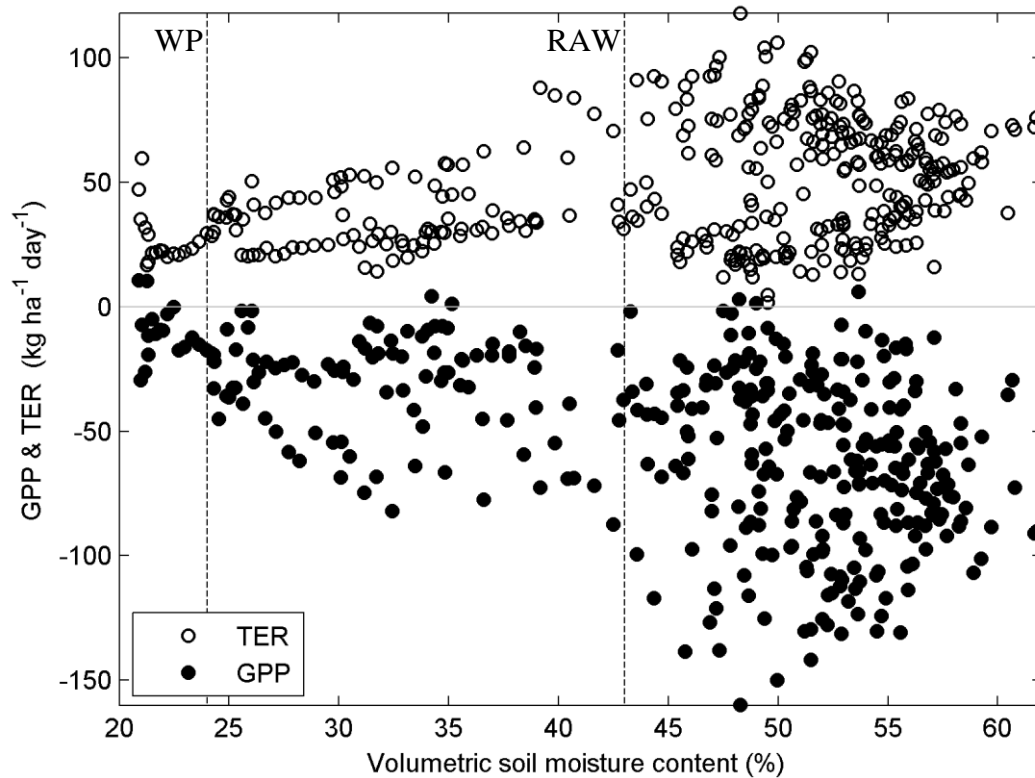
A scatter plot of all valid 30 minute NEE values vs. soil moisture suggests a critical threshold for NEE at a moisture content of about 43% (Fig. 4.3). When soil moisture was below 43%, NEE was generally between 5  $\mu\text{mol m}^{-2} \text{s}^{-1}$  and  $-10 \mu\text{mol m}^{-2} \text{s}^{-1}$ , while when soil moisture was above 43% the range of NEE values was much greater with values generally being between 15  $\mu\text{mol m}^{-2} \text{s}^{-1}$  and  $-30 \mu\text{mol m}^{-2} \text{s}^{-1}$ . There appears to be a second break point at a soil moisture content of about 24%, below which net photosynthesis practically stopped (Fig. 4.3). These breakpoints correspond well to likely water retention properties of the Matangi soil. The Matangi silt loam is found adjacent to the Bruntwood, Te Kowhai and Horotiu soils, with all these soils grading into one another. For the Bruntwood, Te Kowhai and Horotiu soils, Singleton, (1991) showed that 40 – 45% was the lower limit of readily available water ( $-100 \text{ kPa}$ ), below which plants begin to become moisture stressed, and 23 – 25 % was the lower limit of total available water (i.e.  $-1500 \text{ kPa}$ , the wilting point).

There were a few large positive peaks in NEE when soil moisture was less than 43% (including one time when soil moisture was only about 21%). These peaks coincided with rainfall events and will be explored further in Section 4.3.2.2).

Daily totals of GPP and TER showed the same general pattern as 30 minute NEE (Fig. 4.4). When soil moisture was <43% the majority of GPP and TER values were less than 50 kg C ha<sup>-1</sup> day<sup>-1</sup>, but when soil moisture was greater than 43%, GPP values of more than 100 kg C ha<sup>-1</sup> day<sup>-1</sup> were reasonably common and TER was often between 50 and 100 kg C ha<sup>-1</sup> day<sup>-1</sup> (Fig. 4.4). However, when soil moisture was above 43% there were still quite a few daily values for GPP and TER that were <50 kg ha<sup>-1</sup> day<sup>-1</sup>. These low values occurred during winter when soil temperatures were low (Fig. 4.2). Maximum GPP and TER occurred when soil moisture content was around 50%. The daily results for GPP and TER indicate that the decrease in NEE (increased C uptake) observed when soil moisture was greater than 43% (Fig. 4.3) was caused by a proportionally greater increase of GPP than of TER. The increase in TER as soil moisture increased would have been due to increased heterotrophic decomposition of soil C and probably increased root respiration.



**Fig. 4.3.** Average 30 minute net ecosystem exchange of CO<sub>2</sub> (NEE) vs. volumetric soil moisture content for the entire dataset. The vertical dotted lines represent the lower limit of readily available water (RAW) and the wilting point (WP). RAW and WP thresholds derived from Singleton, (1991) for soils similar to the Matangi Silt loam.

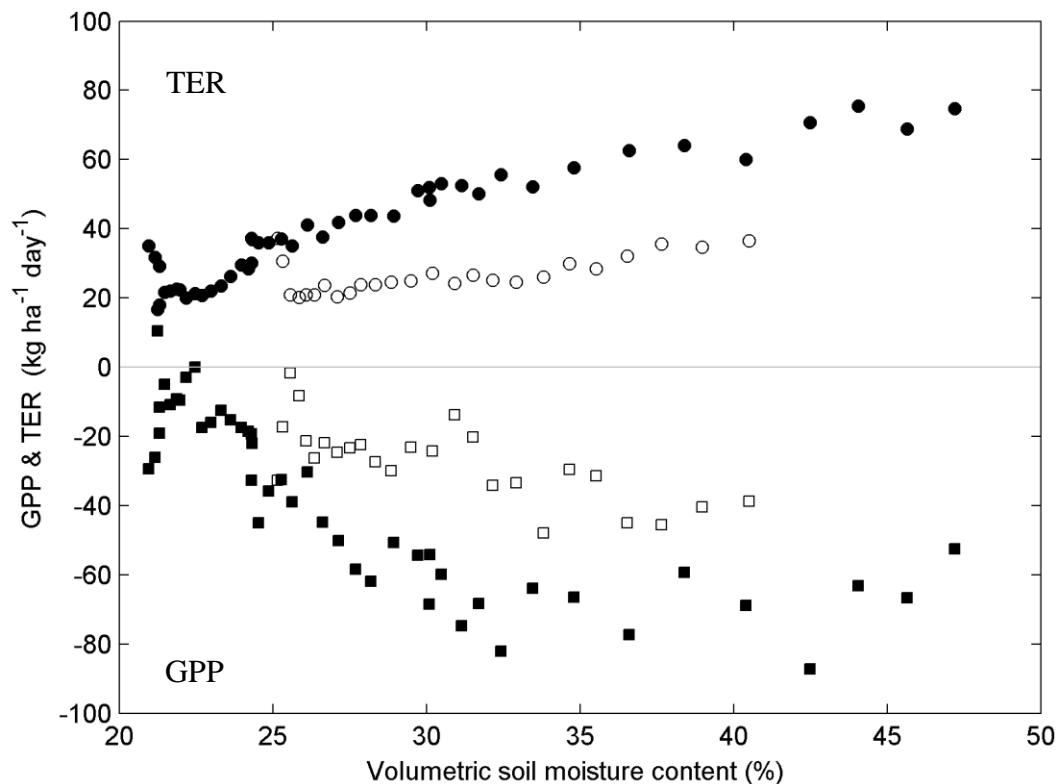


**Fig. 4.4.** Daily totals of modelled total ecosystem respiration (TER) and gross primary production (GPP, plotted as negative values) vs. daily mean volumetric soil moisture content. The dotted vertical lines represent the lower limit of readily available water (RAW) and the wilting point (WP). RAW and WP thresholds derived from Singleton, (1991) for soils similar to the Matangi silt loam.

When soil moisture was below 50%, daily totals of TER and GPP were clumped into two distinct groups (Fig. 4.4). Closer inspection revealed that these two groups could be separated based on time, with the majority of values corresponding to two drying periods when soil moisture was constantly declining. From 26 December 2007 to 9 February 2008 there was very little rain and the soil dried rapidly from about 46% to 21%. Both TER and GPP also declined rapidly during this period (Fig. 4.5). It was interesting to note that GPP only began declining when soil moisture was below about 33% while TER declined over the whole period (Fig. 4.5). Between 10 February and 4 March there were intermittent rain events, with the largest events on 12 February and between 1 – 4 March causing soil moisture to increase to about 40%. The second major drying period occurred between 5 March and 28 March. During this period there was virtually no rain and soil moisture content declined from about 40% to 25% (similar range to first drying period) (Fig. 4.5). In this second drying period, TER and GPP

values were much lower than during the first drying period and only declined slightly as the soil dried (Fig. 4.5).

The most likely cause for the difference in GPP and TER between these two periods was that during the drying event from December 2007 to February 2008 plants would have initially been actively growing and not moisture stressed. By the end of January, soil conditions were so dry that many pasture plants had died (approx 50%). Therefore, although soil moisture during the drying period in March was similar to during January and early February, GPP was lower because there were less plants alive to photosynthesise. Lower GPP was also probably one of the main causes for the lower TER during March, because reduced autotrophic respiration and reduced C inputs to the soil via root exudates would have limited microbial respiration (Craine *et al.*, 1999; Luo and Zhou, 2006). Microbial respiration is also reduced in dry soils due to a reduction in the thickness of water films on soil particles which inhibits diffusion of substrates (Davidson and Janssens, 2006) and reduces the volume of water into which microbes can discharge wastes (Cook and Orchard, 2008). Drying can also directly kill microbes (Wu and Brookes, 2005).

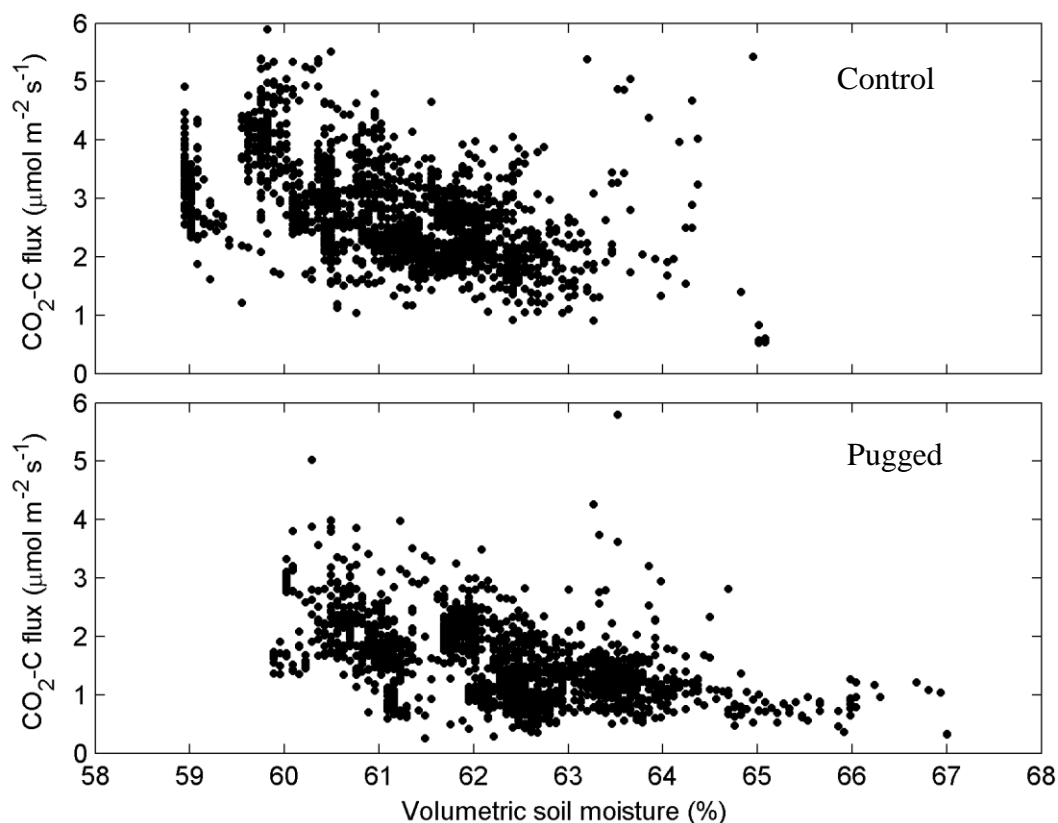


**Fig. 4.5.** Daily totals of modelled total ecosystem respiration (TER) and gross primary production (GPP, plotted as negative values) vs. daily mean volumetric soil moisture content. Closed symbols represent data from the first drying period (26 December 2007 – 9 February 2008) and open symbols data from the second drying period (5 March – 29 March). During both these periods the soil was constantly drying and there was little rain (see Fig. 4.2).

The fourfold decline in respiration between 26 December and 9 February (Figs. 4.2, 4.5) as the soil dried from near field capacity (~50%, -10 kPa) to below wilting point (~24%, -1500 kPa) was similar to a number of other grassland studies. Hunt *et al.* (2002) found that soil respiration declined by a factor of four as the soil dried from field capacity (21%, -10 kPa) to the lowest recorded value of only 4%, in a tussock grassland in southern New Zealand. In an annual grassland in California, Xu *et al.* (2004) observed that TER began to decline when soil moisture fell below 15% (-800 kPa) and then practically ceased when volumetric soil moisture content was below 10% (-2000 kPa). Soil moisture also limited respiration in a tallgrass prairie, where Franzluebbers *et al.* (2002) found a significant increase in respiration as soil moisture increased in the range 20 – 60%. This trend was only significant if soil temperature was  $\geq 10$  °C. In a warm temperate grassland, Novick *et al.* (2004) found that soil moisture contents as low as ~12% had only a minor effect on TER, despite low soil moisture causing a decline in GPP due to reduced stomatal conductance. In a laboratory study, Fang and Moncrieff (2001) found no difference in respiration rates between intact soil cores (from farmland and forest), with volumetric moisture contents ranging from 20 to 50% (similar range to the current study). However, comparing CO<sub>2</sub> exchange results between studies, in terms of absolute values for volumetric soil moisture is of limited value. This is because the biological processes of photosynthesis and respiration are not controlled so much by soil moisture content, but rather by the energy state of the water (i.e. matrix potential), which is determined by the combination of soil moisture content and pore size distribution.

While decreasing soil moisture contents decreased TER, there was also evidence that increasing soil moisture contents, when soil moisture was already high, could decreased respiration. During the pugging trial, measurements made with the automatic chamber revealed that there was a general decline in respiration rates as soil moisture increased (Fig. 4.6). Soil moisture content next to the automatic chamber collars was always >59% during the pugging trial. Total porosity of the Matangi soil at the pugging trial site was about 66% (calculated from bulk density of 890 kg m<sup>-3</sup> and assuming particle density of 2,650 kg m<sup>-3</sup>), which indicated that the soil was near saturation much of the time and heavy rain events would

have caused the soil to become fully saturated (see Fig. 4.10). The decline in respiration with increasing soil moisture was probably caused by reduced microbial activity as a result of low oxygen concentrations due to slow diffusion through water rather than air (Fang and Moncrieff, 1999; Davidson and Janssens, 2006).



**Fig. 4.6.** CO<sub>2</sub> flux and volumetric soil moisture content measured every 15 minutes with the automatic chamber in a control plot and pugged plot of the pugging trial from 25 June to 5 August 2008.

#### 4.3.2.2 Short term effects of rain and soil moisture on CO<sub>2</sub> exchange

At annual and monthly time scales, soil moisture was one of the key factors controlling CO<sub>2</sub> exchange at Scott Farm. Soil moisture content is driven by inputs from rainfall and losses from evaporation and so not surprisingly, rainfall had a large effect on CO<sub>2</sub> exchange, which was most obvious on short time scales (e.g. minutes-days).

##### Respiration pulses following rain

Rainfall events during the drought period (when soil moisture <43%) usually caused large, though relatively short lived (days) increases in NEE (loss of C to

the atmosphere) (Fig. 4.7). These increases in NEE were most likely caused by increases in respiration rather than a decrease in photosynthesis. This suggestion was supported by data from chamber measurements made during the cultivation trial, which showed a peak in respiration (from pasture paddocks) after rain on 10 February of similar magnitude to the peak in NEE measured by the EC system (see Chapter 3, Section 3.3.3, Fig. 3.7). Numerous EC studies have reported large increases in respiration following rain events when soils were previously dry (Hunt *et al.*, 2004; Xu and Baldocchi, 2004; Xu *et al.*, 2004; Aires *et al.*, 2008; Baldocchi, 2008). Chamber studies in the field (Jarvis *et al.*, 2004; Bauer *et al.*, 2006) and laboratory incubations (Birch, 1959; Fierer and Schimel, 2003; Wu and Brookes, 2005) have also shown pulses of respiration following wetting of dry soil. This effect has become known as the “Birch effect” after the pioneering work of H.F. Birch in East Africa in the 1950s and 1960s (Jarvis *et al.*, 2004). A number of mechanisms have been proposed to explain this effect with the traditional view being that wetting releases physically protected organic material, and that previous drying killed some of the microbial biomass making a large pool of labile C available for microbial decomposition following rewetting (Birch, 1959; Wu and Brookes, 2005). Another proposed mechanism is that on re-wetting microbes release cytoplasmic solutes to maintain cell water potentials and these highly labile substances are subsequently mineralized (Fierer and Schimel, 2003; Jarvis *et al.*, 2004). The Birch Effect has predominantly been observed in desert and semi arid regions or seasonally dry ecosystems such as savannahs. It is possible that in most ‘normal’ years in the Waikato this effect may not occur and was only observed in this study because of the extreme drought.

At Scott Farm, cumulative C loss between 10 February and 17 February (i.e. timing of first rain events after drought) was 315 kg ha<sup>-1</sup>, which was 17% of annual NEE. Large pulses of respiration following rain (on previously dry soil), have also had an important influence on annual C budgets in other studies (Hunt *et al.*, 2004; Jarvis *et al.*, 2004; Xu and Baldocchi, 2004). The large C losses during these short pulse events, emphasises the need for continuous measurements (Xu *et al.*, 2004), especially since current models are not able to accurately predict CO<sub>2</sub> losses during these events (Hunt *et al.*, 2004; Jarvis *et al.*, 2004). There is concern that if climate patterns change, with more intense rainfall and more

frequent droughts, C may be lost from soils, because wetting and drying cycles tend to cause greater C losses than from soils that are constantly moist (Jarvis *et al.*, 2004).

The large losses of CO<sub>2</sub> measured by both the EC system and two chamber systems following rain events during the drought period would have been due to decomposition of above ground pasture material and/or soil organic matter. Evidence for a contribution from above-ground dead material to respiration peaks after rain events during the summer/autumn was that respiration pulses occurred immediately after rain, even when there was insufficient rain to increase soil moisture at 50 mm (the depth of the first soil moisture sensor) (Fig. 4.5). This indicated that a large amount of respiration was derived from dead plant material on the soil surface and/or from the surface layer of soil.

To determine whether above ground material could have contributed significantly to the large losses of CO<sub>2</sub> after rain events, a simple experiment was conducted. Grass in the automatic chamber collar in the pasture paddock of the cultivation trial was lightly sprayed with water (the soil was not wet). Wetting resulted in an almost instantaneous increase in respiration from about 2.5  $\mu\text{mol m}^{-2} \text{s}^{-1}$  (25 kg C ha<sup>-1</sup> day<sup>-1</sup>) to 7  $\mu\text{mol m}^{-2} \text{s}^{-1}$  (73 kg C ha<sup>-1</sup> day<sup>-1</sup>) (Fig. 4.8). Due to the hot sunny and windy conditions at the time, the grass dried rapidly which caused a relatively fast decline in respiration rate and after about an hour the respiration rate had declined to 3.4  $\mu\text{mol m}^{-2} \text{s}^{-1}$  (35 kg C ha<sup>-1</sup> day<sup>-1</sup>). A second application of water caused a similar increase in respiration (Fig. 4.8). Results were also presented in kg C ha<sup>-1</sup> day<sup>-1</sup> to allow a direct comparison with daily NEE (Fig. 4.7). Daily NEE increased by about 50 kg C ha<sup>-1</sup> day<sup>-1</sup> after the first major rain event on 10 February. This suggests that decomposition of above-ground dead pasture material could have made a major contribution to the increase in respiration observed following rain events, during the drought period. Using chamber methods, Madsen *et al.* (2007) also observed large increases in respiration following rain from collars containing dead crop residue. To test whether the increase in CO<sub>2</sub> emissions was due to decomposition of surface residue, they incubated crop residues in the lab and measured the respiration rate. When the residue was dry, respiration rates were near zero, but when the residue was wetted



the respiration rate increased rapidly (and of a similar magnitude to in the field). They suggested that following rain, a large proportion of respiration measured in the field was derived from crop residues.

At the end of the cultivation trial (5 March), above ground dead plant biomass within collars from the pasture paddocks was 1,414 kg DM ha<sup>-1</sup>, (707 kg C ha<sup>-1</sup>, assuming 50% C). Dead plant biomass in these paddocks was similar to the paddocks surrounding the EC system. Between 6 March and 14 April, cumulative TER calculated from EC measurements was 1,212 kg C ha<sup>-1</sup>, which indicated that there was enough above-ground material present on the soil surface to account for about half of the observed respiration during this period. However, not all of the dead pasture material had disappeared by mid April.

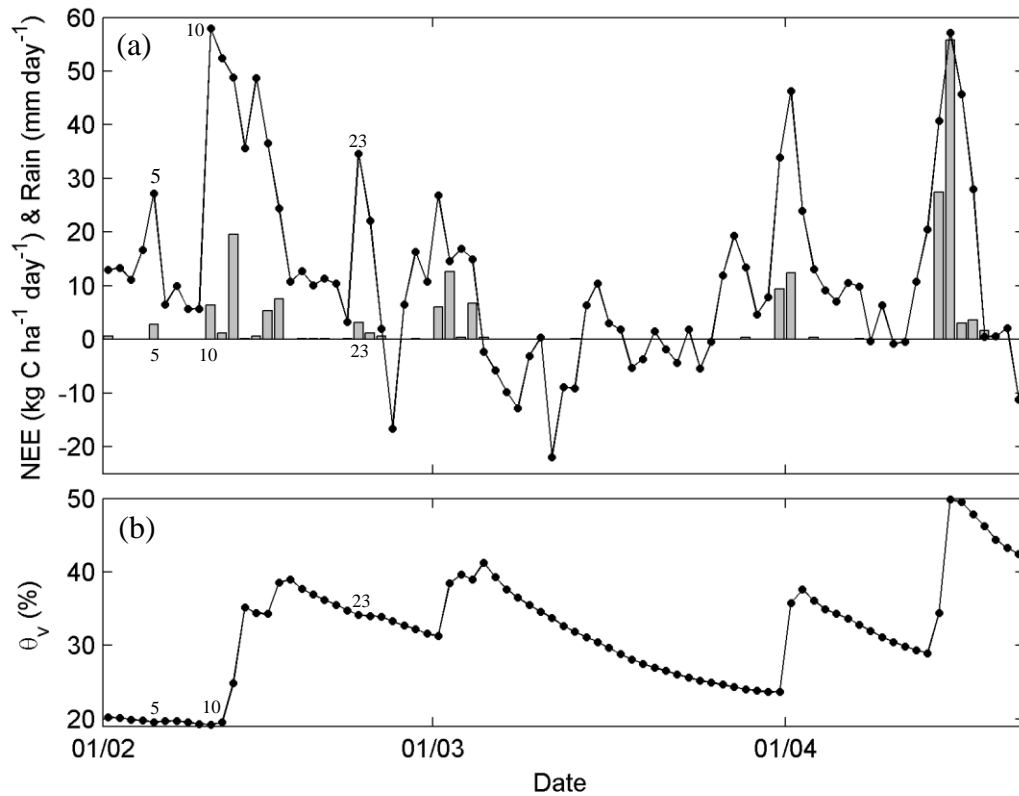


Fig. 4.7. (a) Daily totals of net ecosystem exchange of CO<sub>2</sub> (NEE) and sum of daily rain (bars) during the drought (1 February 2008 to 22 April 2008) and (b) daily mean 50 mm soil moisture content ( $\theta_v$ ) over the same period. The small numbers 5, 10, and 23 are days in February when rainfall caused an increase in NEE but no appreciable increase in soil moisture.

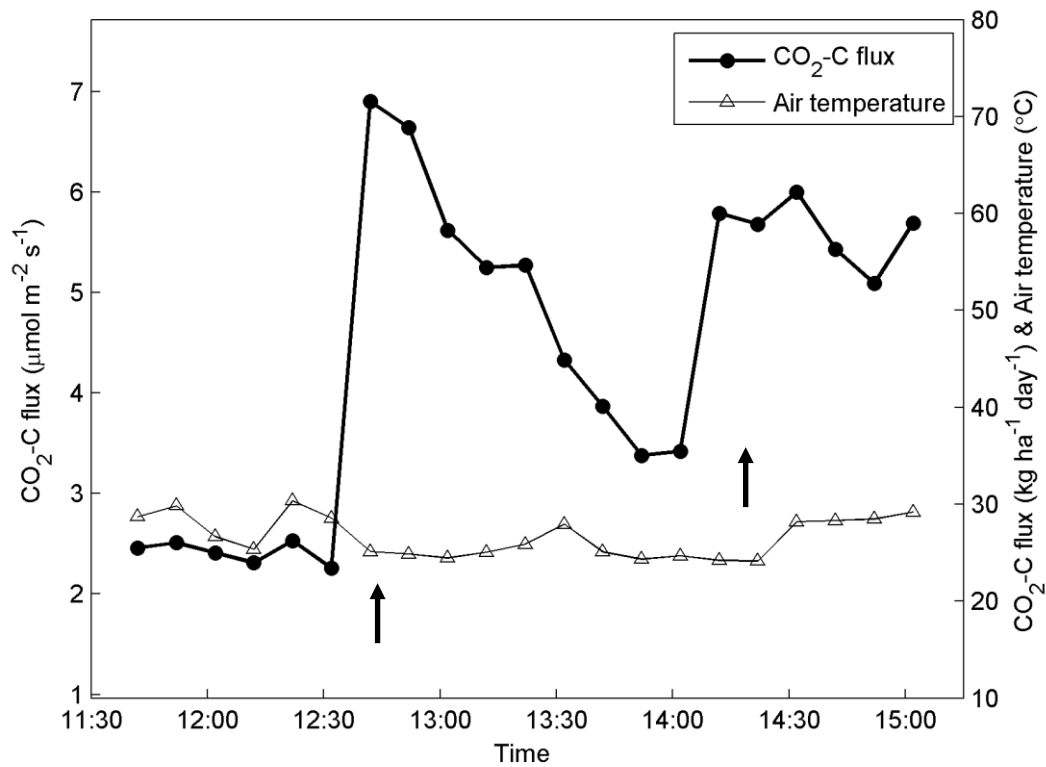
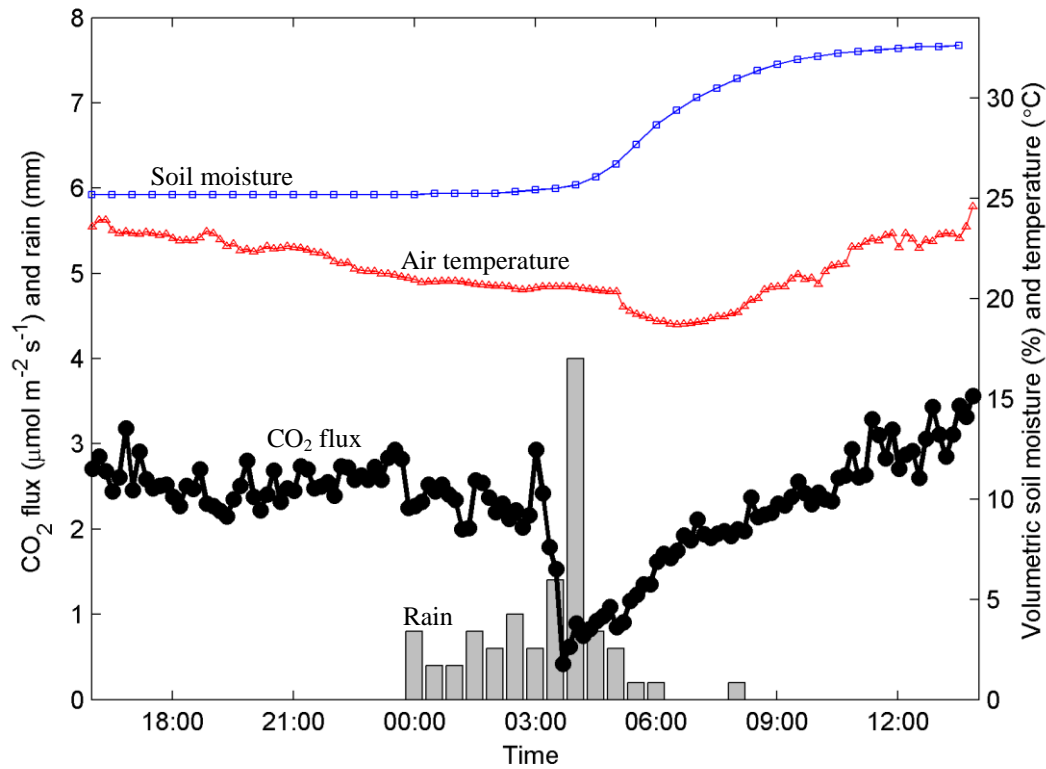


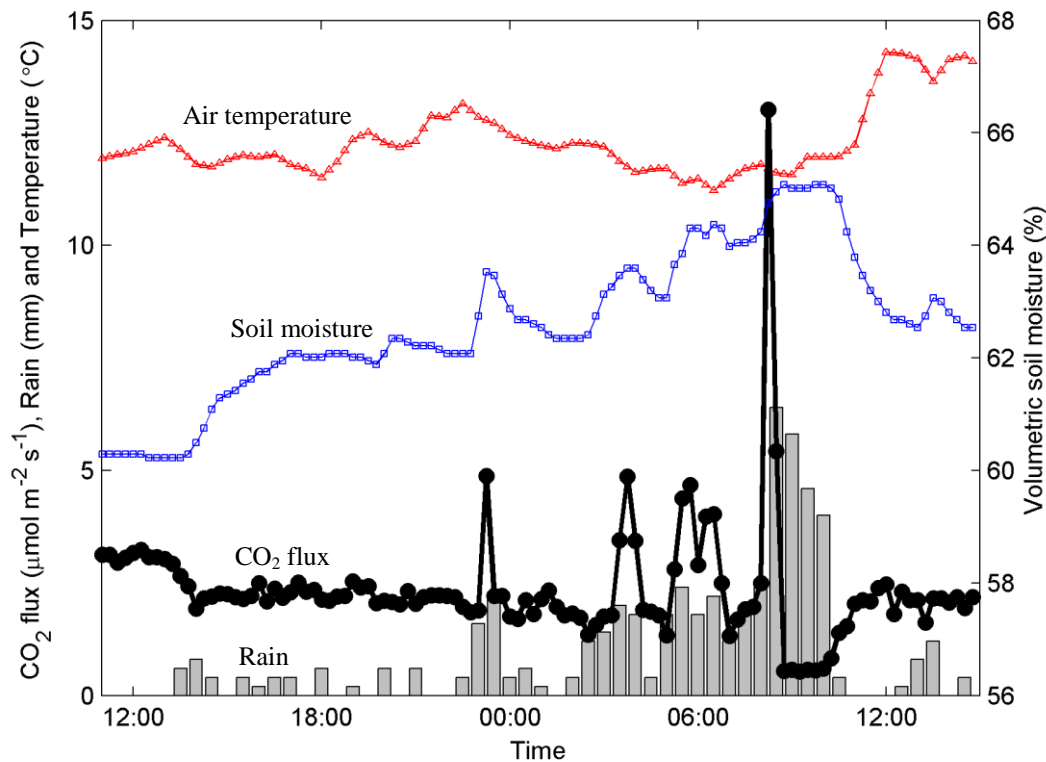
Fig. 4.8. CO<sub>2</sub>-C flux and air temperature measured with the automatic chamber from the pasture paddock of the cultivation trial on 10 March. Arrows indicate when the grass within the collar was lightly sprayed with water (soil was not wet). Measurements were made at 10 minute intervals. The right y axis is in kg C ha<sup>-1</sup> day<sup>-1</sup> to allow comparison with Fig. 4.7.

*Surface sealing and mass flow*

The previous section documented that rain during the drought period generally resulted in large increases in CO<sub>2</sub> emissions. However, rain did not always cause large increases in CO<sub>2</sub> emissions and in some cases caused marked decreases. Close inspection of automatic chamber measurements from the cultivation study revealed three instances where heavy rain coincided with a sharp decrease in CO<sub>2</sub> emissions (twice in the cultivated paddock and once in the pasture paddock). An example from in the cultivated paddock is shown in Fig. 4.9. Similar decreases in CO<sub>2</sub> emissions were also observed after rain during the pugging trial, (in both pugged and control plots). However, during the pugging trial heavy rain events sometimes caused large, very short term flushes of CO<sub>2</sub>. An example from the control plot in the pugging trial from 29 and 30 July is shown in Fig. 4.10. This example was chosen because both depression of fluxes due to rain and increases due to rain occurred within 24 h. Between 11:00 and 14:00 on 29 July, there was no rain and CO<sub>2</sub> emissions were about 3  $\mu\text{mol m}^{-2} \text{s}^{-1}$ . Light intermittent rain commenced after 14:00, soil moisture increased slightly but fluxes decreased slightly to 2.5  $\mu\text{mol m}^{-2} \text{s}^{-1}$  remaining relatively constant until 23:00. Between 23:00 on 29 July and 06:00 on 30 July rainfall intensity increased and there were three peaks in soil moisture which coincided with rapid increases in CO<sub>2</sub> emissions to about 5  $\mu\text{mol m}^{-2} \text{s}^{-1}$ . At 08:30 on 30 July very heavy rain commenced, soil moisture increased further and the CO<sub>2</sub> flux increased to 13  $\mu\text{mol m}^{-2} \text{s}^{-1}$ . CO<sub>2</sub> fluxes then dropped rapidly to 5.4  $\mu\text{mol m}^{-2} \text{s}^{-1}$  and then fluxes for the following 1.5 hours were only 0.5  $\mu\text{mol m}^{-2} \text{s}^{-1}$ . Soil moisture during this time remained constant at about 65%, very close to saturation (total porosity was 66%). Soil moisture then declined rapidly to about 62.5% and then levelled off, which was mirrored by CO<sub>2</sub> emissions which increased and then plateaued again at about 2.5  $\mu\text{mol m}^{-2} \text{s}^{-1}$  (Fig. 4.10).



**Fig. 4.9.** CO<sub>2</sub> flux (filled circles) and air temperature (open triangles) measured every 10 minutes with the automatic chamber in the cultivated paddock of the cultivation trial on 1 and 2 March, along with the sum of 30 minute rainfall (bars) and 30 minute average volumetric soil moisture content (open triangles).



**Fig. 4.10.** CO<sub>2</sub> flux (filled circles), volumetric soil moisture content (open squares), and air temperature (open triangles) measured every 15 minutes with the automatic chamber in the control plot of the pugging trial on 29 and 30 July, along with the sum of 30 minute rainfall (bars). Soil total porosity was 66% (i.e. all soil pores full of water at volumetric moisture content of 66%).

The sharp increases in CO<sub>2</sub> emissions following some rain events were unlikely to have been due to increased soil moisture content alleviating moisture stress of microbes or plants, because soil moisture content was always above 60%. These large increases in CO<sub>2</sub> emissions were probably caused by water infiltrating into the soil displacing air in the soil which contains high concentrations of CO<sub>2</sub> (Reicosky *et al.*, 1999; Eriksen and Jensen, 2001; Luo and Zhou, 2006). These large short lived increases in CO<sub>2</sub> emissions following rain that occurred during the pugging trial were less common and lasted a much shorter time than the depression of fluxes due to rain. These short-term fluxes of CO<sub>2</sub> would be difficult to accurately predict because both biological processes and mass flow of CO<sub>2</sub> would need to be modelled.

Ball *et al.* (1999) also observed a large decrease in CO<sub>2</sub> emissions for about a day following heavy rain on a no-till field in Scotland, and suggested that the decrease was due to anaerobic conditions limiting respiration. This suggestion was supported by the observation of increased N<sub>2</sub>O emissions during the same period of time. Madsen *et al.* (2007) measured CO<sub>2</sub> emissions with 16 automatic chambers in a corn/soybean field in Nebraska and found that rain increased CO<sub>2</sub> emissions from collars with large amounts of crop residue but decreased emissions from collars with bare soil. They attributed the increase in emissions from collars with residue to the “Birch effect” (discussed above) and the decrease in emissions from collars without residue to reduced gas transport in the surface soil. The very high water filled pore space after rain events in the current study suggests that both anaerobic conditions or reduced gas transport could have contributed to the decrease in CO<sub>2</sub> emissions that occurred after some rain events.

#### Implications of short term CO<sub>2</sub> fluxes due to rain

During rain, water on the IRGA windows of the eddy covariance system prevented accurate measurements and data during rain events were rejected. Therefore the large short-term increases or decreases in CO<sub>2</sub> emissions that coincided with rain events would not have been measured by the EC system, potentially biasing results. This problem was likely compounded because gaps in the data caused by rain were filled using data from non-rain periods. Falge *et al.*

(2001) warned of this problem and suggested that process-orientated gas exchange models should be used in cases where gaps were biased towards a particular condition. However, the ability of such models to accurately predict the varied effects that rain has on CO<sub>2</sub> emissions is likely to be limited. In wet environments, closed path EC systems may be better than open path systems, because they can continue to make measurements during rain (Gilmanov *et al.*, 2007).

#### Summary of soil moisture and rain effects

At annual time scales, the usual expected effects of soil moisture on respiration and photosynthesis were observed. There appeared to be a critical soil moisture threshold at about 43% (the lower limit of readily available water), with both respiration and photosynthesis declining when soil moisture was below this threshold. When soil moisture was below the wilting point (~23%), TER and GPP were always very low, with GPP practically ceasing. During the drought period, rainfall events generally coincided with large pulses of respiration, a large proportion of which may have been derived from surface plant litter. During the pugging trial, when soils were very wet, heavy rainfall events occasionally coincided with large peaks in CO<sub>2</sub> emissions, most likely caused by water displacing air in the soil, which contains high concentrations of CO<sub>2</sub>. However, heavy rain sometimes caused a sharp decrease in CO<sub>2</sub> emissions, probably because anaerobic conditions limited soil respiration or diffusion of CO<sub>2</sub> from the soil. These data support the conceptual model that changes in soil moisture content effects on CO<sub>2</sub> production are greatest at low soil moisture content (<45%) and high (approaching saturation) soil moisture contents (Fang and Moncrieff, 1999; Davidson *et al.*, 2000; Fang and Moncrieff, 2001; Luo and Zhou, 2006).

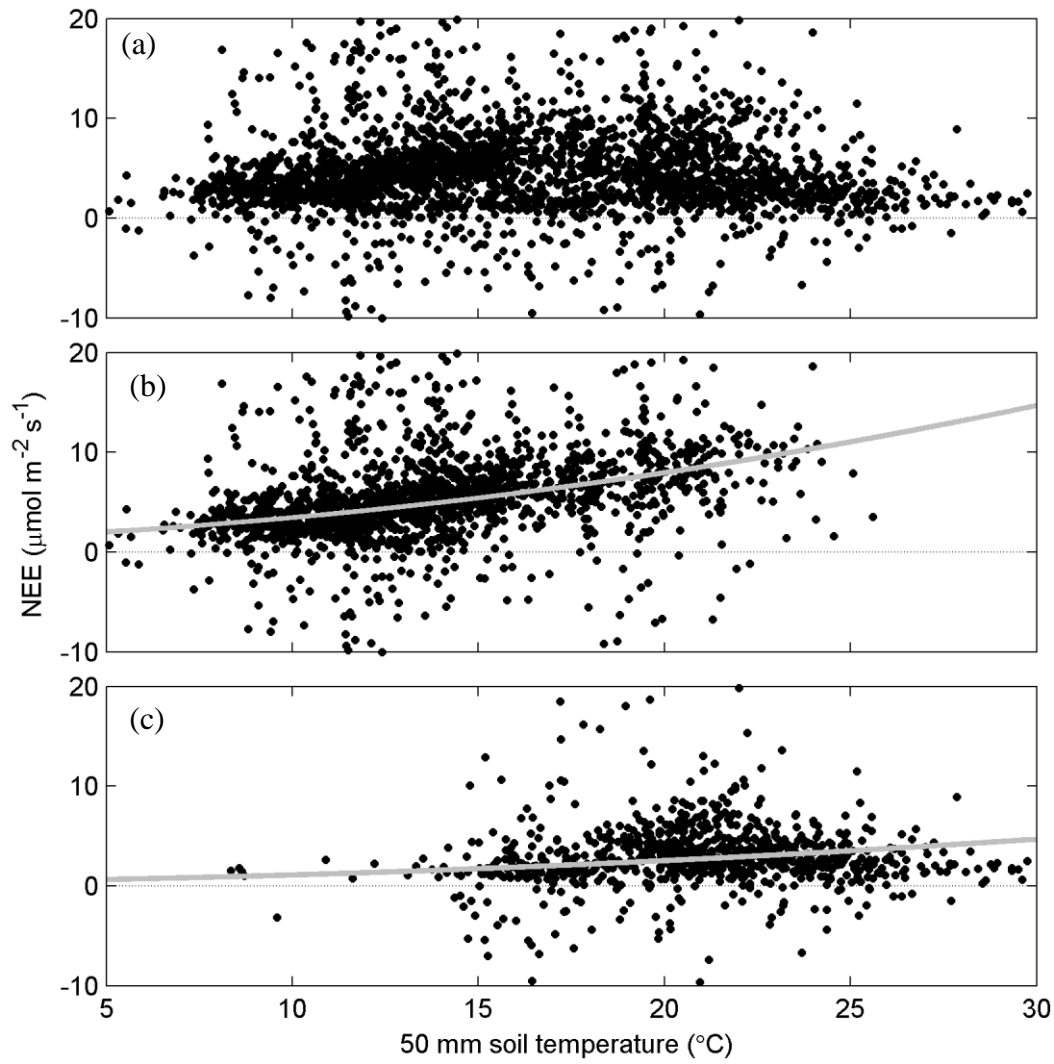
#### **4.3.3 The effect of temperature on CO<sub>2</sub> exchange**

In recent decades, there has been considerable interest in the effect that rising global temperatures might have on rates of respiration and photosynthesis (Davidson and Janssens, 2006). This interest has been primarily driven by the quest to understand the effect that changes in global temperature may have, or are having on the global carbon cycle.

Night time NEE measurements made with the EC system were a direct measure of TER due to the absence of photosynthesis. In general, respiration is considered to increase with increasing temperature (Fang and Moncrieff, 2001; Davidson and Janssens, 2006). However, a plot of all 30 minute night time NEE values vs. soil temperature showed no clear trend of increasing NEE with increasing temperature (Fig. 4.11a). This was in agreement with Xu and Baldocchi (2004) who conducted a study in a Californian annual grassland (subject to summer drought). In both studies, soil moisture limited respiration at warmer temperatures, and soil moisture was inversely correlated to soil temperature, obscuring the temperature–CO<sub>2</sub> flux relationship (Xu and Baldocchi, 2004). In contrast, a range of other studies have observed a clear increase in NEE with increasing temperature when all 30 minute NEE values are plotted. For example, a pasture on peat in the Waikato (Nieveen *et al.*, 2005), peat bogs in the Waikato (Smith, 2003; Thornburrow, 2005) and managed grasslands in Europe (Jaksic *et al.*, 2006; Ammann *et al.*, 2007).

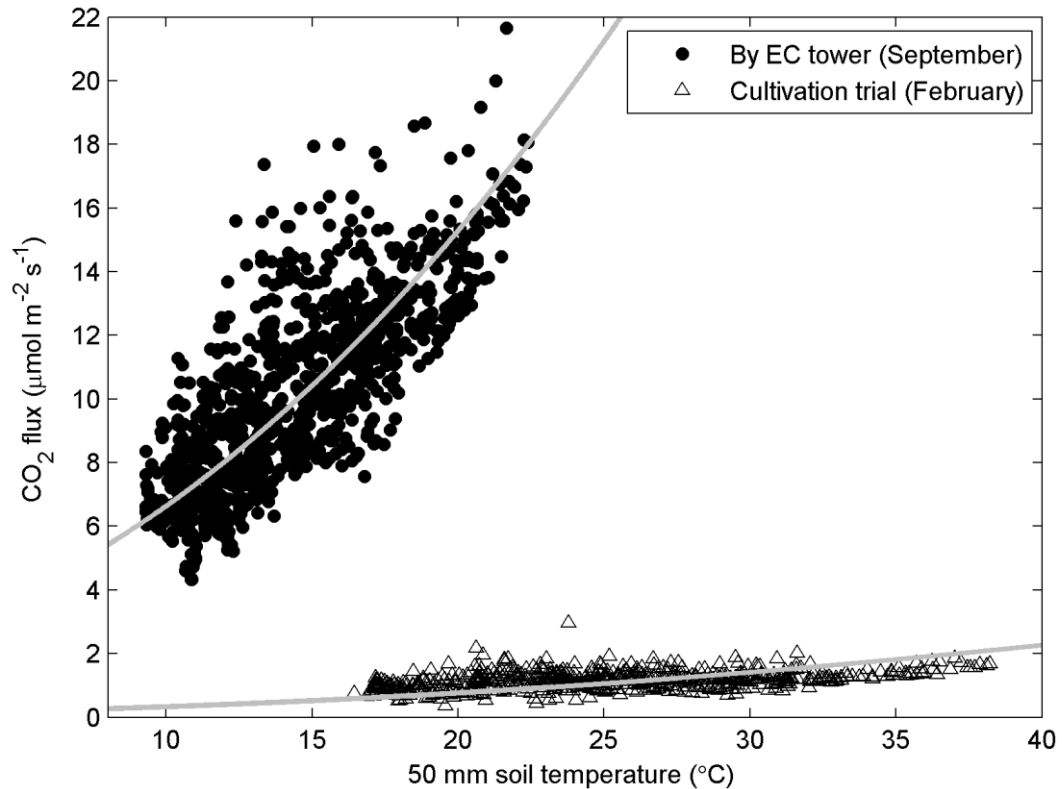
The confounding influence of moisture on temperature can be isolated by separating night time NEE data from Scott Farm into two groups based on soil moisture ( $>43\%$  and  $\leq 43\%$ ). When soil moisture was  $>43\%$  there was a reasonably clear increase in night time NEE with increasing 50 mm soil temperature (Fig. 4.11b), while when soil moisture was  $\leq 43\%$  there was no clear relationship between NEE and temperature (Fig. 4.11c).

Automatic chamber measurements in the pasture paddocks throughout the year also demonstrated that temperature control of respiration was dependent on adequate soil moisture content. Prior to rain on 10 February there was only a very small increase in respiration with increasing soil temperature (Fig. 4.12). In contrast, there was a strong positive correlation between temperature and respiration for automatic chamber measurements made in the control plot during the pugging trial in June, July and August and adjacent to the EC tower in August and September (Fig. 4.12).



**Fig. 4.11.** Average 30 minute night time net ecosystem exchange of CO<sub>2</sub> (NEE) vs. 50 mm soil temperature at Scott Farm. Panel (a) shows all the data, panel (b) data from the non-drought period (volumetric soil moisture >43%) and panel (c) data from the drought period (volumetric soil moisture  $\leq$ 43%). The solid grey lines represent the Lloyd and Taylor (1994) model (equation 4.1) fitted to the data.





**Fig. 4.12.** Respiration vs. 50 mm soil temperature measured with the automatic chamber at Scott Farm next to the EC tower from 12 September – 21 September 2008 (non-drought, 15 minute fluxes) and from the pasture paddock of the cultivation trial from 27 January – 9 February 2008 (drought, 10 minute fluxes). The soil grey lines represent the Lloyd and Taylor (1994) respiration model (equation 4.1) fitted to the data.

It is commonly accepted that respiration is positively correlated with temperature (Fang and Moncrieff, 2001; Davidson and Janssens, 2006) if other factors are not limiting. Night time NEE data and automatic chamber data from Scott Farm showed that respiration increased with increasing temperature only when soil moisture was >43% (the lower boundary of readily available water). The soil moisture threshold of about 43% at Scott Farm was similar to the 40% threshold reported by Franzluebbers *et al.* (2002). They found that there was no significant increase in whole ecosystem or soil respiration with an increase in temperature when soil moisture was <40%, but a significant increase when soil moisture was  $\geq 40\%$ .

Respiration at 10 °C ( $R_{10}$ ) derived from equations such as Lloyd and Taylor (1994) is a commonly used metric to compare between studies. The  $R_{10}$  at Scott Farm calculated from EC data during the non-drought period was  $3.44 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$  (Fig. 4.11b).  $R_{10}$  during the drought period was only  $1.1 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ ,

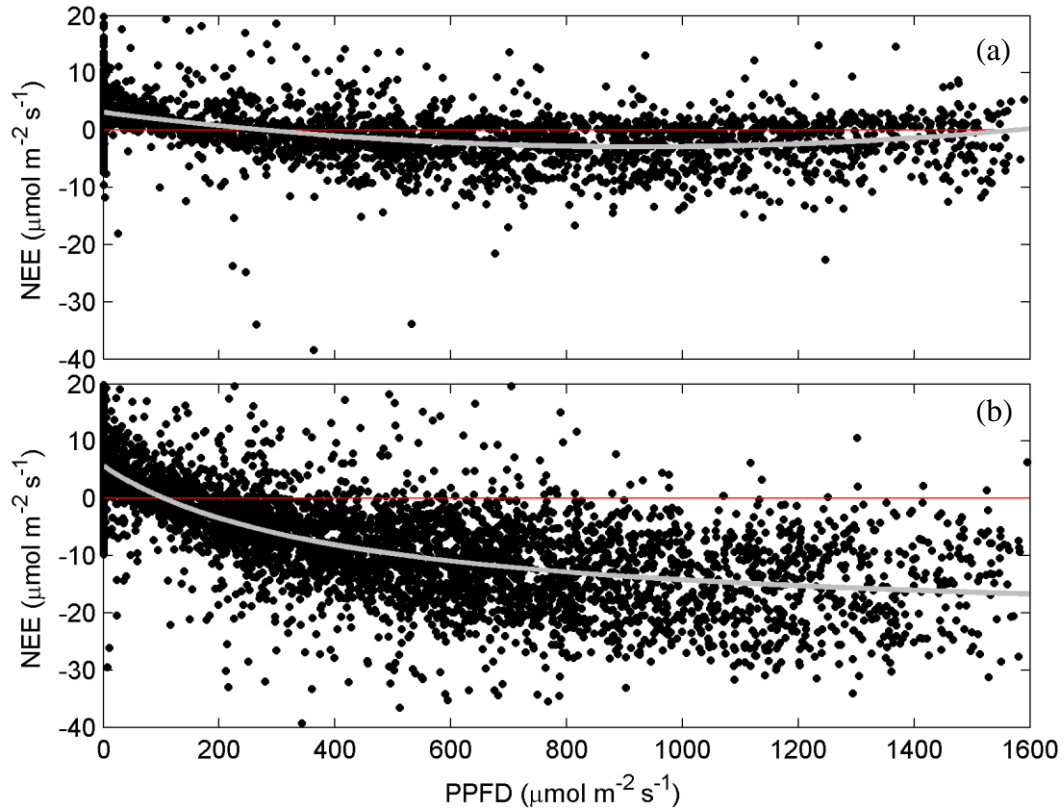
but this value should be treated with caution because there were few temperatures around 10 °C. The  $R_{10}$  of 3.44  $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ , during the non-drought period, was higher than the value of 2.44  $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$  reported by Nieveen *et al.* (2005) for a whole year for a similar farm system in the Waikato, but on peat soil. For a pasture system on a mineral soil in the South Island of New Zealand, Brown *et al.* (2009) calculated a much higher  $R_{10}$  of 4.77  $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$  from a 250 day study.

#### 4.3.4 The effect of light on CO<sub>2</sub> exchange

Because light drives photosynthesis it is often the key control on CO<sub>2</sub> exchange in pasture systems. Photosynthetic photon flux density (PPFD) has been shown to explain 80 – 95% of variation in grassland NEE during short (1 – 15 day) time periods (Xu and Baldocchi, 2004; Gilmanov *et al.*, 2007; Aires *et al.*, 2008). Factors which affect plant growth, such as soil moisture limitation or grazing and harvesting can affect the relationship between PPFD and NEE (Nieveen *et al.*, 2005).

At Scott Farm, the relationship between 30 minute NEE values and photosynthetic photon flux density (PPFD) (called a NEE light response curve (Gilmanov *et al.*, 2007)) was strongly affected by soil moisture. When soil moisture was >43%, the NEE light response curve showed a typical pattern of increasing C uptake (decreasing NEE) with increasing PPFD (Fig. 4.13b) (Novick *et al.*, 2004; Jaksic *et al.*, 2006; Ammann *et al.*, 2007; Gilmanov *et al.*, 2007). When soil moisture was ≤43% the light response curve was much flatter, with only a small increase in C uptake as PPFD increased (Fig. 4.13a). The much flatter NEE light response curve during the drought was consistent with other studies (Xu and Baldocchi, 2004; Nieveen *et al.*, 2005; Aires *et al.*, 2008), and was most likely caused by decreased photosynthetic uptake of CO<sub>2</sub> because plants were moisture stressed. On a dairy farm in Ireland, Jaksic *et al.* (2006) demonstrated that NEE light response curves during August (summer), for two years were almost identical, despite soil moisture contents being quite different (average of about 24% and 35% for the two years respectively). They explained that this similarity occurred because although soil moisture was considerably different, plants were not

moisture stressed in either year (the wilting point of their soil was 12%, and total soil porosity was 49%).

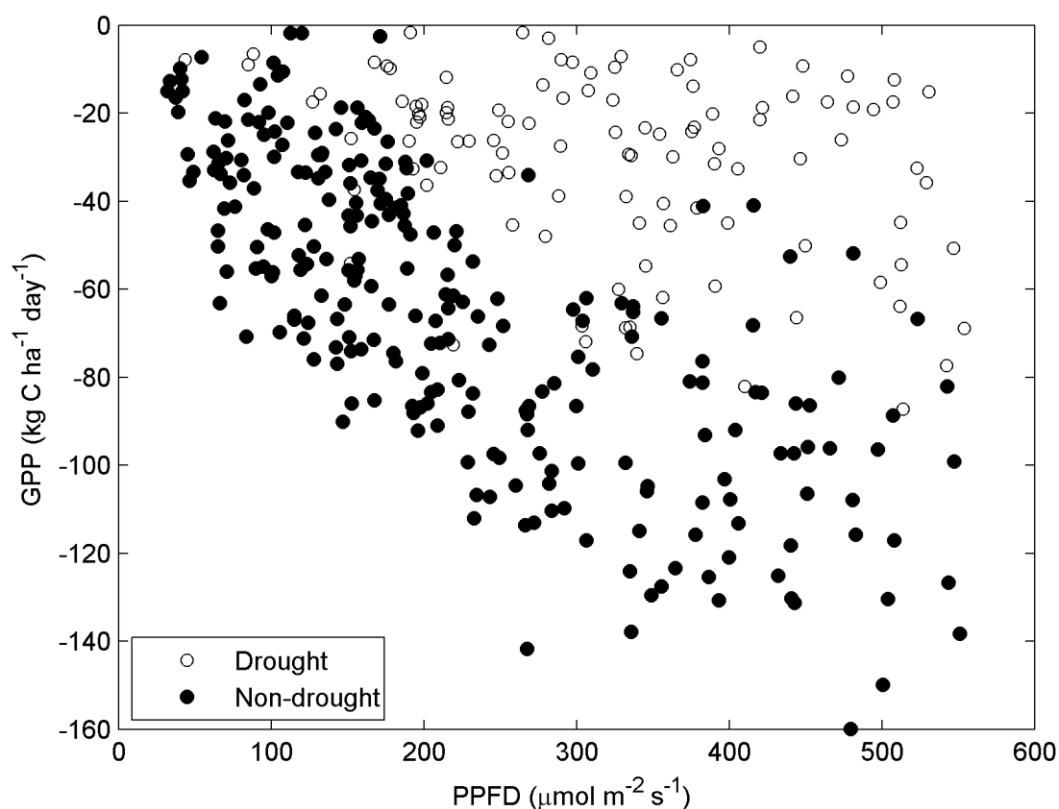


**Fig. 4.13.** Average 30 minute net ecosystem exchange of CO<sub>2</sub> (NEE) vs. photosynthetic photon flux density (PPFD) at Scott Farm, (a) during the drought (volumetric soil moisture  $\leq 43\%$ ) and (b) during the non drought period (volumetric soil moisture  $> 43\%$ ). The grey line in (a) was fitted with a 2<sup>nd</sup> order polynomial ( $r^2 = 0.25$ ) and the grey line in (b) was fitted with the Michaelis-Menten rectangular hyperbola ( $r^2 = 0.62$ ) (Aires *et al.*, 2008).

Most studies that investigate the relationship between NEE and PPFD use short 1 – 15 day time windows to avoid confounding effects of changing soil moisture, temperature and vegetation (Gilmanov *et al.*, 2007; Aires *et al.*, 2008). Light response curves were not fitted to data for such short time windows in this study, but a comparison of coefficients from the rectangular hyperbola (equation 4.2) fitted to the data from the non-drought period (Fig. 4.13b), reveals that results from this study were similar to other studies.  $GPP_{max}$  was  $28.2 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ , which falls within the range of values reported by Aires *et al.* (2008) ( $-9.4$  to  $-36.9$ ) and Xu and Baldocchi (2004) ( $-10.8$  to  $-40.2$ ) for grasslands in Mediterranean climates. In an un-grazed temperate grassland in Canada, the highest  $GPP_{max}$  for three years was only  $-27.5 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ , and this was using only data from non-cloudy days during the period of peak photosynthetic activity (Flanagan *et al.*, 2002). The apparent quantum yield for data from the

non-drought period was 0.067, which was higher than any other values found in the literature (Suyker and Verma, 2001; Flanagan *et al.*, 2002; Xu and Baldocchi, 2004; Aires *et al.*, 2008). K was 420  $\mu\text{mol m}^{-2} \text{s}^{-1}$  and TER 5.6  $\mu\text{mol m}^{-2} \text{s}^{-1}$ .

Total daily GPP was dependent on daily mean PPFD during the non-drought period where an increase in mean PPFD resulted in a large increase in GPP (Fig. 4.14). In contrast, there was only slight increase in GPP with increasing PPFD during the drought period. Not unexpectedly, this indicated that in the absence of moisture limitation, the greater the total amount of light (due to longer days or greater intensity) the more C plants fixed.



**Fig. 4.14.** Total daily gross primary production (GPP, plotted as negative values) vs. daily mean photosynthetic photon flux density (PPFD) for the drought (soil moisture  $\leq 43\%$ ) and non-drought period (soil moisture  $> 43\%$ ).

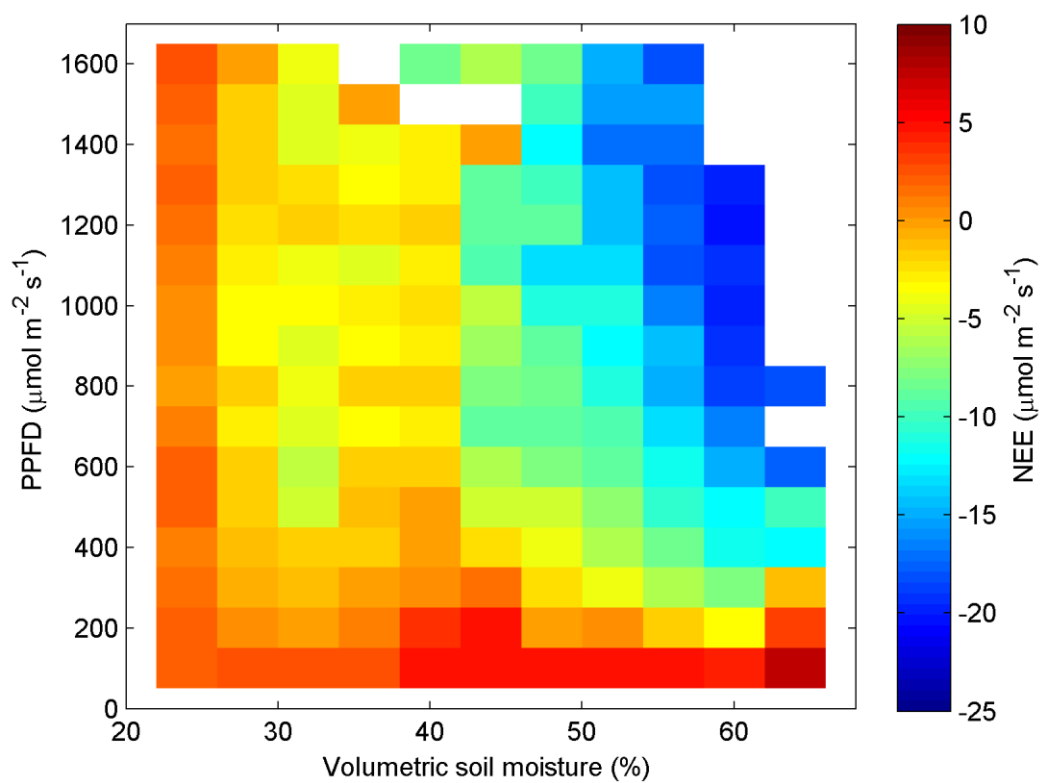
#### 4.3.5 Interactions between temperature, moisture, PPFD and NEE

From the above analysis, it was clear that the environmental drivers of temperature, moisture and PPFD each had an effect on NEE. In order to obtain a general understanding of the interaction between these three variables and NEE, pseudocolour plots were constructed. These plots allowed one variable (e.g.

temperature) to be plotted on the x axes and another (e.g. PPFD) on the y axes, with the magnitude of NEE displayed as different colours. To construct these figures, NEE data (filtered, but un-gapfilled) that fell within specified ranges of the other variables (2 °C bins for temperature, 4% bins for soil moisture and 100  $\mu\text{mol m}^{-2} \text{s}^{-1}$  bins for PPFD) were averaged. If there were less than five valid NEE measurements within the intersect between two bins (e.g. temperature and PPFD) these combined bins were treated as empty. Using this approach it was also possible to directly examine the controls of TER after filtering out daytime data.

#### Soil moisture PPFD and NEE

When PPFD was  $<100 \mu\text{mol m}^{-2} \text{s}^{-1}$  (i.e. night time and change of light), the soil-pasture system as expected was always a C source (positive NEE) (Fig. 4.15). The soil-pasture system was also a C source (or zero) when soil moisture was  $< 26\%$  at all levels of PPFD. This soil moisture threshold was similar to the lower limit of plant available water (wilting point) of about 24% reported by Singleton (1991) for soils similar to the Matangi silt loam. When soil moisture was between 26 – 42% the soil-pasture system was generally a weak C sink when PPFD was above about  $300 \mu\text{mol m}^{-2} \text{s}^{-1}$  although sink intensity did not increase with increasing PPFD. A clear change occurred when soil moisture was above 42% with much greater C uptake and an increase in C uptake as PPFD increased (Fig. 4.15). This clear breakpoint is consistent with the lower limit of readily available water reported by Singleton (1991). Sink intensity was greatest when soil moisture was above 50% and PPFD above  $500 \mu\text{mol m}^{-2} \text{s}^{-1}$  (Fig. 4.15).

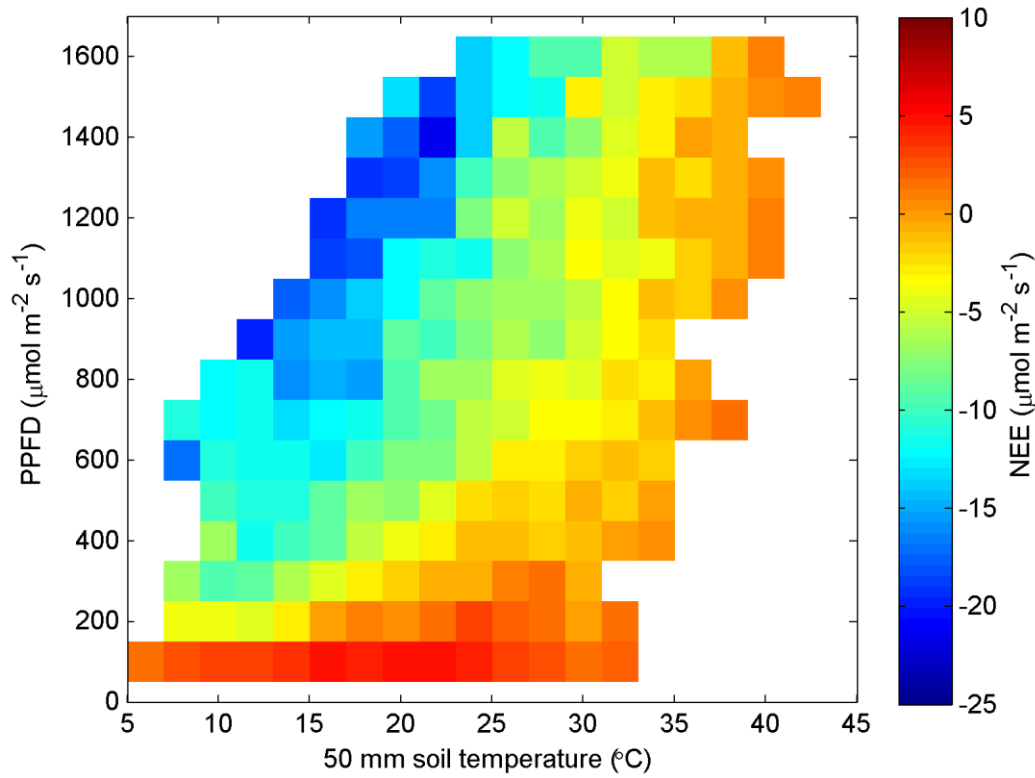


**Fig. 4.15.** Pseudocolour graph showing the relationship between photosynthetic photon flux density (PPFD), volumetric soil moisture content and binned averages of 30 minute net ecosystem CO<sub>2</sub> exchange (NEE).

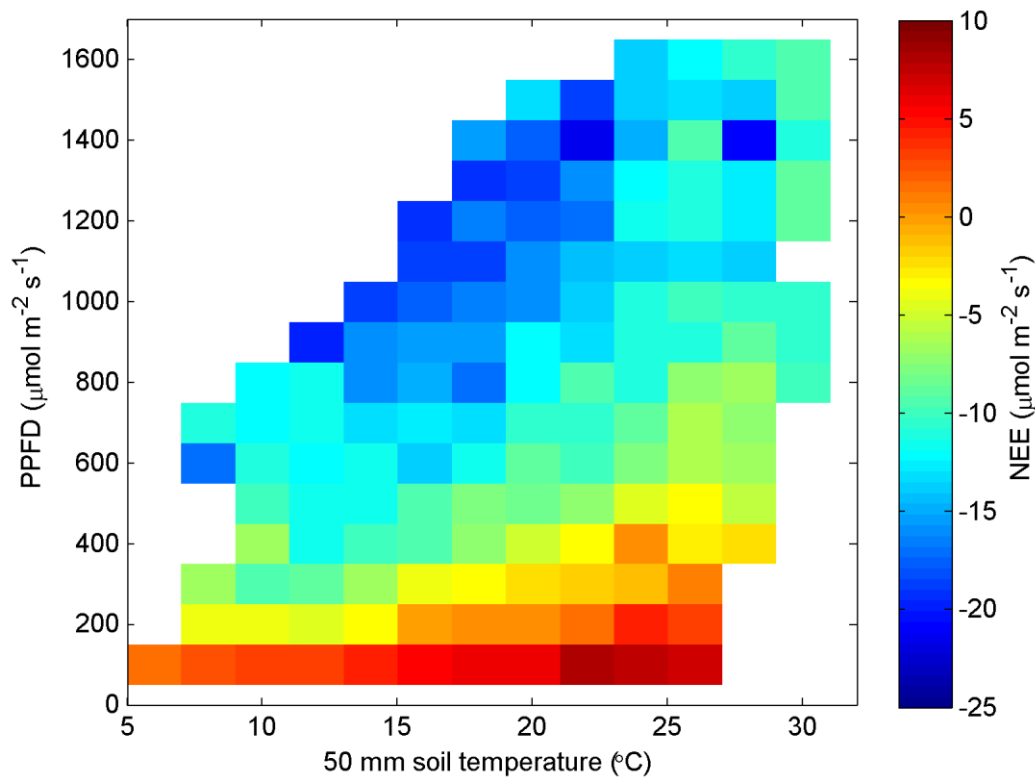
PPFD, soil temperature and NEE

When soil temperature was held steady, an increase in PPFD led to increased C uptake (more negative NEE), except when temperatures were greater than 37 °C (Fig. 4.16). The exception when soil temperatures were >37 °C was probably because soil temperatures were only this high during the drought period when soil moisture limited photosynthesis.

When PPFD was above 100  $\mu\text{mol m}^{-2} \text{s}^{-1}$ , C uptake decreased (more positive NEE) as soil temperature increased (Fig. 4.16). This trend was largely due to the fact that most of the higher soil temperatures occurred during the drought period when soil moisture limited photosynthesis. However, when data from the drought period were excluded, the same trend was still evident, although not as strong (Fig. 4.17). This suggests that factors other than drought were contributing to this trend of decreasing C uptake with increasing temperature at a constant PPFD. One simple explanation for this effect is that respiration increased proportionally more than photosynthesis as temperature increased (Xu and Baldocchi, 2004). Gilmanov *et al.* (2007) demonstrated that the light response curves of grasslands can exhibit hysteresis, where net C uptake for a given PPFD is greater in the morning (when temperatures are cool) than in the afternoon (when temperatures are warmer).



**Fig. 4.16.** Pseudocolour graph showing the relationship between photosynthetic photon flux density (PPFD), 50 mm soil temperature and binned averages of 30 minute net ecosystem CO<sub>2</sub> exchange (NEE).



**Fig. 4.17.** Pseudocolour graph showing the relationship between photosynthetic photon flux density (PPFD), 50 mm soil temperature and binned averages of 30 minute net ecosystem CO<sub>2</sub> exchange (NEE), for the non-drought period (volumetric soil moisture >43%).

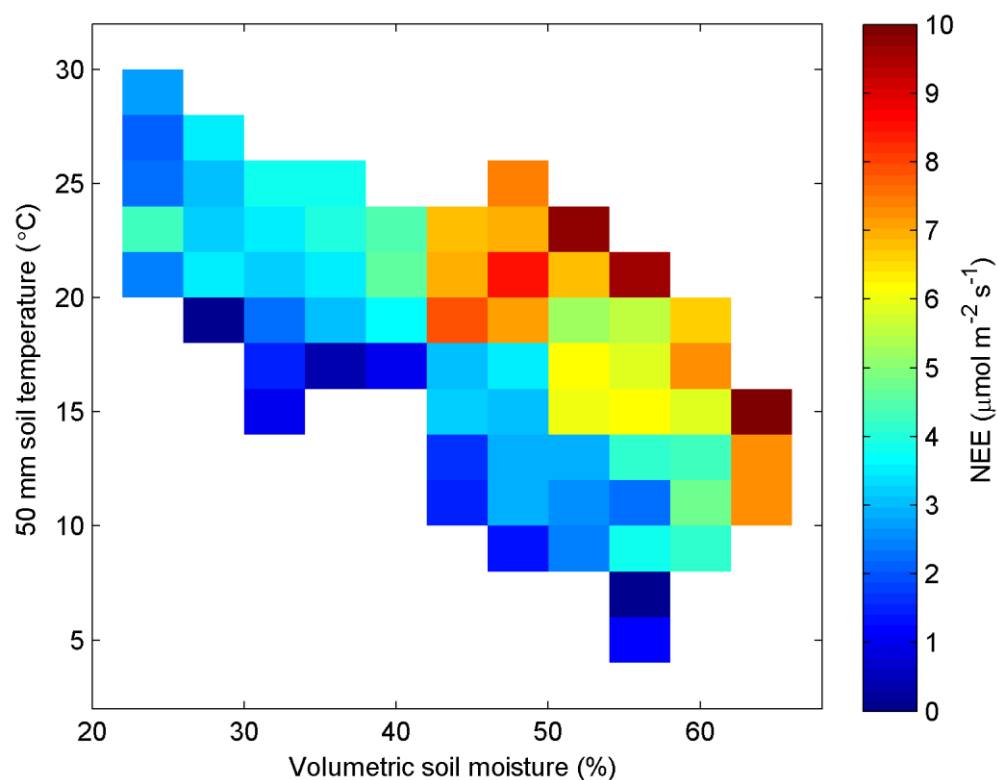


*Night time NEE and daytime NEE vs. soil moisture and temperature*

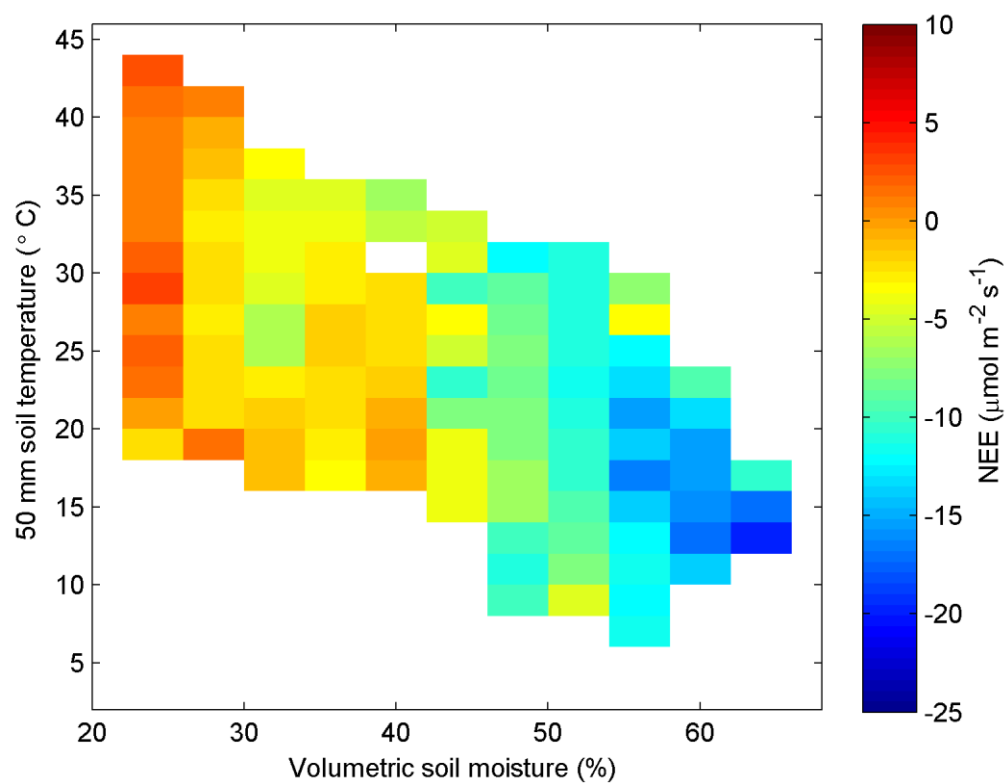
At night, NEE was a direct measure of total ecosystem respiration because photosynthesis had stopped. When soil moisture was below 42% bin averaged respiration rates were low ( $<5 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ ) and there was only a minor increase in respiration with increasing temperature (Fig. 4.18). In contrast, when soil moisture was above 42%, bin averaged respiration rates were as high as  $10 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$  and there was a relatively large increase in respiration as soil temperature increased. For daytime data, there were no clear trends of increasing C uptake with increasing soil temperature across all soil moisture contents (Fig. 4.19). During the day, an increase in soil moisture resulted in a general increase in C uptake, with soil moisture contents of about 26% and 42% appearing to be critical thresholds above and below which the magnitude of NEE changed (Fig. 4.19).

*Summary of interactions between soil temperature, moisture, PPFD and NEE*

The pseudocolour plots (4.15 – 4.19) showed the complex interactions between different environmental drivers (moisture content, PPFD and temperature) and NEE. These interactions clearly demonstrate that modelling NEE (or component parts, TER and GPP) based simply on PPFD and soil temperature will be difficult in environments where seasonal changes in soil moisture has a large influence on CO<sub>2</sub> exchange. Soil moisture content is dependent on rainfall, but soil physical properties (total porosity and pore size distribution), will determine the availability of this water and these properties vary between soils.



**Fig. 4.18.** Pseudocolour graph showing the relationship between volumetric soil moisture content, 50 mm soil temperature and binned averages of 30 minute night time net ecosystem CO<sub>2</sub> exchange (NEE).



**Fig. 4.19.** Pseudocolour graph showing the relationship between volumetric soil moisture content, 50 mm soil temperature and binned averages of 30 minute daytime net ecosystem CO<sub>2</sub> exchange (NEE).

## 4.4 Conclusions

The severe drought had a marked impact on CO<sub>2</sub> exchange at Scott Farm. Respiration and photosynthesis were both reduced when volumetric soil moisture content was below 43% (~the lower limit of readily available water) and photosynthesis virtually ceased when soil moisture declined below 24% (~wilting point).

Soil moisture also had a large influence on the relationship between temperature and respiration. When soil moisture was >43%, respiration increased as soil temperature increased, while when soil moisture was below 43% there was no clear increase in respiration with temperature. The relationship between PPFD and NEE was also strongly affected by soil moisture. When soil moisture was above 43%, the NEE light response curve showed a typical decrease in NEE (increased C uptake) with an increase in PPFD, while when soil moisture was below 43% the light response curve was much flatter.

Rain and soil moisture also had a large influence on CO<sub>2</sub> emissions over much shorter time periods. Rain events during the drought period (soil moisture ≤43%) caused large short-term losses of CO<sub>2</sub>. A large proportion of the CO<sub>2</sub> lost during these events probably originated from decomposition of above ground plant litter, rather than soil respiration. Occasionally, heavy rain events during the pugging trial caused large, but short-lived, increases in CO<sub>2</sub> emissions. These peaks in CO<sub>2</sub> emissions were attributed to water displacing air from the soil, rather than water alleviating moisture stress because the soil was already very wet during the pugging trial. Automatic chamber measurements revealed that rain events sometimes also caused short term decreases in CO<sub>2</sub> emissions, presumably due to reduced air-filled porosity causing anaerobic conditions which limited soil respiration. These large increases or decreases in CO<sub>2</sub> emissions associated with some rain events would not have been measured by the EC system because data during rain were usually rejected. This could have potentially biased estimates of daily, monthly and annual sums of NEE because these missing data were gap-filled with data from non-rain periods (Falge *et al.*, 2001).

Pseudocolour plots were a useful approach to explore the complex interactions between soil temperature and moisture, PPFD and NEE and confirmed many of the conclusions (e.g. soil moisture thresholds) derived from using the traditional approach of scatter plots and non-linear regression. One effect that the pseudocolour plots clearly demonstrated, that was not shown with other analyses, was that for a set PPFD range, an increase in soil temperature resulted in a decrease in C uptake. Most of this effect was caused because the higher soil temperatures occurred during the drought period when soil moisture limited photosynthesis. However, another contributing factor may have been that respiration increased proportionally more than photosynthesis as temperature increased (Xu and Baldocchi, 2004).

Further work at Scott Farm trying to identify the effects of management such as grazing and silage harvesting on CO<sub>2</sub> exchange is warranted. However, this will be difficult because of the multiple small paddocks within the flux footprint all grazed/harvested at different times. Therefore more complex footprint analysis will be required to confirm that fluxes have originated from the paddocks of interest.

## 5. SUMMARY AND CONCLUSIONS

### 5.1 Introduction

Soil carbon (C) is important because even small changes in soil C can affect atmospheric concentrations of CO<sub>2</sub> which in turn can influence global climate. Adequate soil carbon is also required to maintain soil quality, which is important if agricultural production is to be sustained.

The soil carbon balance of New Zealand pasture soils is poorly understood, with some studies reporting increases in soil C, some no change and some decreases.

The overall aim of this research was to improve our understanding of the carbon balance of dairy farm soils. An annual carbon balance of an intensive dairy farm in the Waikato, North Island, New Zealand was calculated from CO<sub>2</sub> exchange measurements made with an eddy covariance (EC) system and from farm production data and literature values. CO<sub>2</sub>-C losses following cultivation for pasture renewal in late summer and soil pugging by dairy cattle in winter was quantified using closed chamber techniques. An analysis of environmental controls (temperature, moisture and light) on CO<sub>2</sub> exchange was completed using EC and chamber data.

The next three sections will summarize key results and conclusions in relation to the three objectives of this study presented in Section 1.2. This will be followed by recommendations for further research.

### 5.2 Annual farm scale C budget

When all C inputs and outputs were included, the soil-pasture system at Scott Farm was a net sink of  $-880 \pm 500$  kg C ha<sup>-1</sup> y<sup>-1</sup>. This was similar to the average sink strength of  $-1,040$  kg C ha<sup>-1</sup> y<sup>-1</sup>, reported by Soussana *et al.* (2007), for eight grassland sites in Europe. Annual NEE measured with the eddy covariance system was  $-1,843$  kg C ha<sup>-1</sup> y<sup>-1</sup> (a C sink), despite severe drought conditions during the first part of the study. This was in contrast to other studies which showed that

grasslands were a CO<sub>2</sub>-C source in years with drought (Flanagan *et al.*, 2002; Hunt *et al.*, 2004; Aires *et al.*, 2008). This suggests that Waikato soil-pasture systems may be more resilient to the effects of drought than other grasslands, presumably because the warm temperate climate allows pastures to grow all year and therefore C losses during drought can be recovered during non-drought periods. The large C export in milk of 835 kg C ha<sup>-1</sup> y<sup>-1</sup> highlighted the importance of accounting for product removal when calculating C budgets for New Zealand dairy farms.

The annual farm scale C sink of  $-880 \pm 500$  kg ha<sup>-1</sup> y<sup>-1</sup> was in contrast to the large average soil C losses of 1,060 kg C ha<sup>-1</sup> y<sup>-1</sup>, observed by Schipper *et al.* (2007) at 31 pasture sites (mainly dairy) throughout New Zealand, during the 17 – 30 years prior to 2005. However, more than one year of data is required before the long term C balance of the Scott Farm site is known with certainty.

### 5.3 The effect of physical disturbance on CO<sub>2</sub> emissions

#### 5.3.1 Cultivation

Cumulative CO<sub>2</sub>-C emissions (measured using chambers) from the cultivated and adjacent pasture paddocks, during the 39 day study, were not significantly different (1,496 and 1,446 kg C ha<sup>-1</sup> respectively). However, when C inputs to pasture paddocks via photosynthesis were included in calculations, the net C loss from cultivated paddocks was about 622 kg C ha<sup>-1</sup> more than from adjacent pasture paddocks. This calculated C loss was small in comparison to the annual GPP of  $-19,448$  kg C ha<sup>-1</sup> and TER of  $17,605$  kg C ha<sup>-1</sup>, measured with the eddy covariance system, but was relatively large when compared to the annual farm scale C balance of  $-880 \pm 500$  kg ha<sup>-1</sup> y<sup>-1</sup>. The total C loss during the regrassing process would have been greater than 622 kg ha<sup>-1</sup>, because C loss during the period from spraying to power harrowing (~17 days) was not quantified and pasture establishment was very slow due to dry conditions. The dry conditions during the study also limited soil respiration and C losses from the cultivated paddocks may have been greater if soils were moister.

However, given that cultivation only occurs about every 10 – 15 years in New Zealand pastoral systems, results from the current study suggest that a single cultivation event for pasture renewal is unlikely to cause a large long-term decline in soil C. Cultivation for pasture renewal was therefore unlikely to have contributed significantly to the large C losses observed by Schipper *et al.* (2007).

Further information is needed on the magnitude of CO<sub>2</sub>-C losses following cultivation under wetter conditions and on the rate at which soil C recovers once new pastures become established.

### 5.3.2 *Pugging*

Cumulative CO<sub>2</sub> emissions during the first 41 days following pugging were 1,161 kg C ha<sup>-1</sup> from the pugged plots, which was significantly less than emissions from the control plots (1,381 kg C ha<sup>-1</sup>). Lower emissions from the pugged plots was probably caused by the combination of decreased microbial and root respiration due to anaerobic soil conditions (the pugged plots had consistently higher soil moisture contents than control plots), and also lowered root respiration as a result of lower pasture production. Although the difference in CO<sub>2</sub> emissions between the two treatments was significant, it was probably not an important difference in terms of the long-term soil C balance. However, quantifying the net difference in C loss between the control and pugged plots was not possible because plants were present in both treatments. Total C inputs via photosynthesis would have likely differed between the two treatments (especially since above ground production did).

The main conclusion was that there was not a large loss of CO<sub>2</sub>-C following pugging. However, rainfall during the pugging trial was much higher than usual and a pugging event followed by more ‘normal’ rainfall may have a different effect on CO<sub>2</sub> emissions. It is also not known whether lower pasture production in subsequent months in the pugged soils would lead to lower C inputs to the soil and ultimately lower soil C.

## 5.4 Factors controlling CO<sub>2</sub> exchange

This research occurred during a year with a severe summer drought and a very wet winter, which provided an ideal opportunity to investigate factors controlling CO<sub>2</sub> exchange.

An overall conclusion was that volumetric soil moisture content (hereafter soil moisture) had a dominant effect on CO<sub>2</sub> exchange. Soil moisture contents of 43% and 24% appeared to be critical thresholds, below which both respiration and photosynthesis were progressively reduced. These thresholds were consistent with the lower limit of readily available water (~43%) and the wilting point (~24%), thresholds found by Singleton (1991) for similar soils. Automatic chamber measurements from the pugging trial (when soil moisture was very high) showed that an increase in soil moisture (to near saturation) caused a decline in respiration. Thus, respiration at Scott Farm was reduced at both high and low soil moisture contents, which is consistent with commonly accepted theory (Davidson *et al.*, 2000; Davidson and Janssens, 2006; Luo and Zhou, 2006).

Soil moisture also influenced the relationship between temperature and respiration, and photosynthetic photon flux density (PPFD) and NEE. In the absence of soil moisture limitation (when soil moisture >43%), respiration increased as soil temperature increased. In contrast, there was no clear increase in respiration with temperature when soil moisture was below 43%. C uptake increased rapidly with increasing PPFD when soil moisture was not limiting, while when soil moisture was below 43% an increase in PPFD caused a much smaller increase in C uptake.

At short time scales (i.e. minutes – days) rain and soil moisture also strongly affected CO<sub>2</sub> exchange. Rain events during the drought period (soil moisture ≤43%) generally caused large short-term losses of CO<sub>2</sub> to the atmosphere, which were attributed to the “Birch Effect”. These pulses of respiration were important in terms of the annual C budget, with the cumulative C loss for the first 8 days after the first major rain in February being equivalent to 17% of annual NEE. This large C loss during such a short time period highlighted the need for continuous measurements of CO<sub>2</sub> exchange. Rain during the pugging trial sometimes caused



large short-lived spikes in CO<sub>2</sub> emissions. These spikes were attributed to water displacing air from the soil, rather than increased respiration. Rain sometimes had the opposite effect, causing short-term decreases in CO<sub>2</sub> emissions. This was probably due to reduced air-filled porosity causing anaerobic conditions (which limited soil respiration), or a short-term reduction in the amount of CO<sub>2</sub> diffusing out of the soil. Most of these large, short-term increases or decreases in CO<sub>2</sub> emissions associated with some rain events would not have been measured by the EC system, because data during rain were usually rejected. This would have potentially biased final EC results, especially because gaps were filled with average data from non-rain periods.

## 5.5 Further research

Continued eddy covariance measurements at Scott Farm will allow interannual variability of CO<sub>2</sub> exchange to be assessed. Results from the extreme year of this study could then be compared with a more ‘normal’ year.

Considerably more work could be carried out investigating the controls on CO<sub>2</sub> exchange. For example, identifying the effect of grazing and silage harvest, a more complete analysis of the NEE light response for shorter time periods and an investigation into the effect of vapour pressure deficit on CO<sub>2</sub> exchange. If more years of data were collected from Scott Farm, pseudocolour graphs presented in this thesis could be a powerful tool to explore the interactions between multiple driving factors and NEE. It would be interesting to see if the trends observed in this study are strengthened or blurred with more data.

Eddy covariance studies on other soil types, different pasture systems (e.g. irrigated or drystock) and also in other regions of New Zealand would provide a more complete picture of the carbon balance of New Zealand’s pastoral systems.

Quantifying CO<sub>2</sub>-C emissions following cultivation under wetter (more normal) conditions and on different soil types is required to gain a fuller understanding of the short term CO<sub>2</sub>-C losses following cultivation. Direct drilling following spraying is now a common form of pasture renewal (and crop sowing) and

therefore a comparison between CO<sub>2</sub>-C emissions from cultivated paddocks and sprayed and direct drilled paddocks would be useful, to determine if there is any difference in CO<sub>2</sub>-C loss between these two methods. Often pastures are renewed in conjunction with growing summer crops (e.g. turnips or maize), which requires two cultivation or direct drilling events (spring to sow the crop and autumn to sow new pastures). Quantifying the C loss from this sequence of events is also warranted.

The EC technique may be more suitable than chambers for determining the net CO<sub>2</sub>-C loss that occurs during the pasture renewal/cropping process. This is because EC systems could make continuous measurements of net CO<sub>2</sub>-C exchange throughout the whole process, from spraying, through crop growth and harvest (if a crop was used), until pastures were fully established. It would be ideal if the area surrounding the EC tower was to be cultivated at the end of the current farmlet study, because then comparisons could be made with data from previous years when the EC was over pasture.

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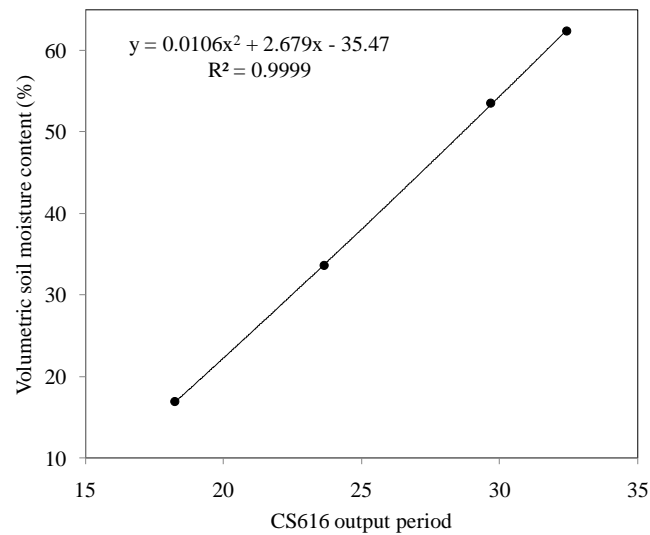
## APPENDIX A: CALIBRATION OF SOIL MOISTURE SENSORS

All soil moisture sensors used in this research (two CS616 and two CS615 water content reflectometers and one CS620 Hydrosense probe, Campbell Scientific Inc, Logan, UT), were calibrated for the Matangi soil in September 2008. The procedure is described below.

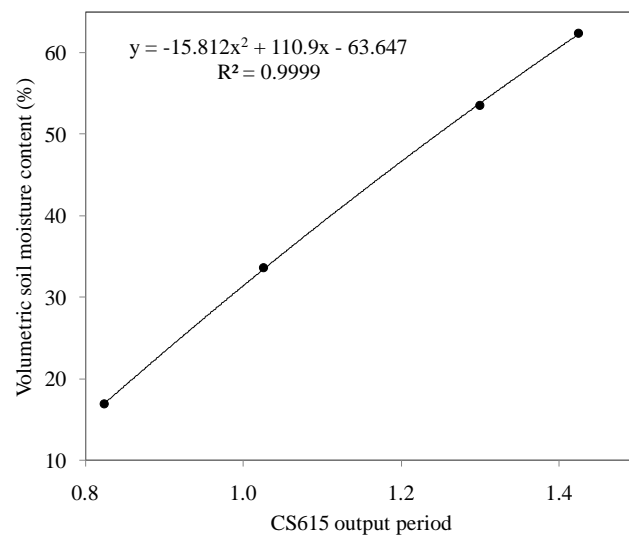
Seven soil cores (98 mm diameter by 75 mm deep) were taken from near the two water content reflectometers installed next to the eddy covariance tower. Cores were taken from 35 – 110 mm depth, to ensure soil was representative of the soil being measured by the two moisture content reflectometers (installed at 50 and 100 mm). Dry bulk density was determined by drying at 105 °C for 48 hours and then weighing.

Additional soil (from the same sites and depths as bulk density cores) was collected and brought back to the laboratory. This soil was sieved to homogenise and then divided into four equal samples (based on weight). One of the four samples was left at the field moist state; while water was applied to another sample and the two other samples were air dried to two different soil moisture contents. This provided four soil samples with volumetric soil moisture contents (17%, 33%, 53% and 62%) spanning the range observed in the field. Each soil sample was then packed into a PVC cylinder (105 mm in diameter and 350 mm long) to the same dry bulk density as in the field ( $770 \text{ kg m}^{-3}$ ).

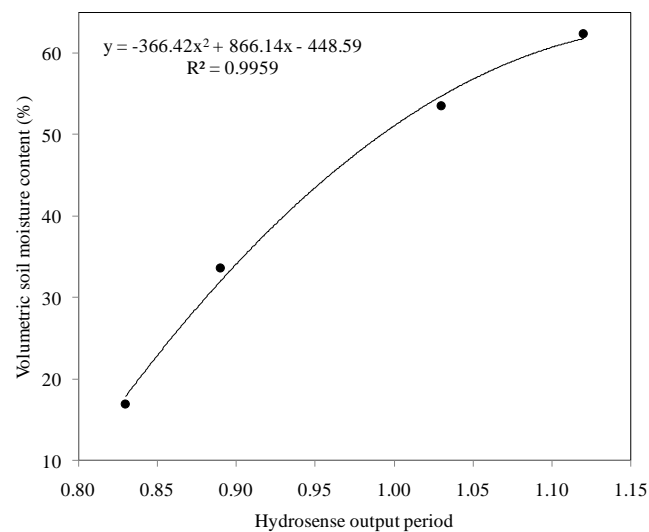
Soil moisture probes were removed from the field and brought back to the laboratory and connected to a datalogger. Each probe was sequentially inserted into each soil column and the output period recorded (signals were recorded every second and then averaged after 1 minute). This procedure was first carried out at room temperature (21 °C) and then the soil columns were packed in ice and placed in a cooler for a few hours (temperature declined to about 10 °C) and the procedure repeated. There was virtually no difference between output periods at the two temperatures and therefore only measurements made at room temperature were used for the calibration. Figures showing the calibration curves and regression equations are presented below.



A1. Calibration curve and equation for the CS616 water content reflectometers located next to the eddy covariance tower.



A2. Calibration curve and equation for the CS615 water content reflectometers used in the cultivation and pugging trials.



A3. Calibration curve and equation for the CS620 Hydrosense probe used in the cultivation and pugging trials.

## APPENDIX B: GAP FILLING AND FLUX PARTITIONING PROCEDURE

Following filtering of the 30 minute net ecosystem CO<sub>2</sub> exchange (NEE) data (as described in Chapter 3), gaps were filled and NEE partitioned between gross primary productivity (GPP) and total ecosystem respiration (TER) using the online software by Reichstein *et al.* (2005). This gap filling approach is one of two ‘standard’ techniques adopted by the Carboeurope IP and FLUXNET initiatives (Moffat *et al.*, 2007). The online software can be found at: <http://gaia.agraria.unitus.it/database/carboeuropeip/> under “data” and then “online gap filling model”. A brief description of the gap filling and flux partitioning procedures is given below. For full details go to the above website or refer to Reichstein *et al.* (2005).

### a) Gap filling

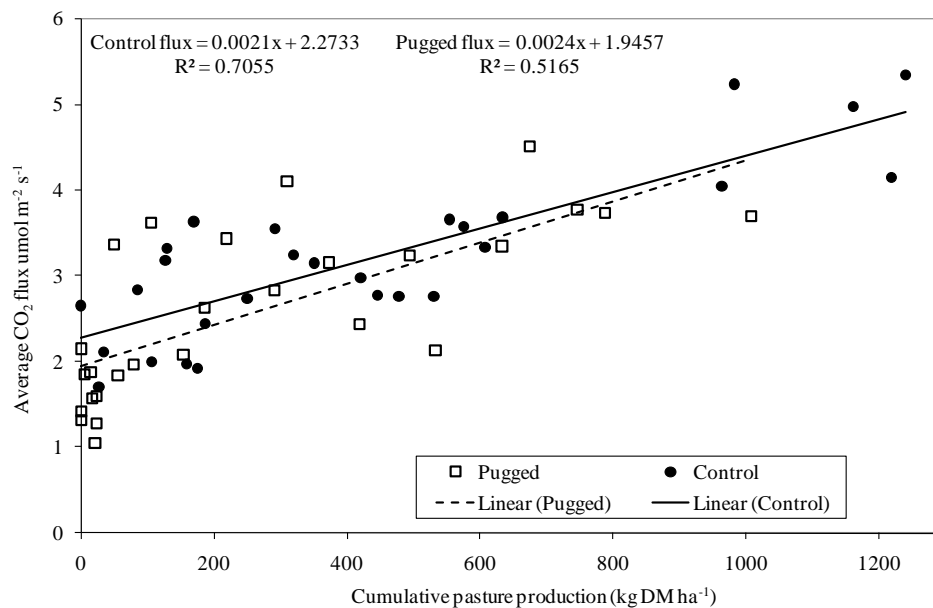
The gap filling procedure was based around the “covariation of fluxes with meteorological variables and the temporal auto-correlation of fluxes” (Reichstein *et al.*, 2005). If NEE data were missing, they were replaced with average NEE values from periods with similar meteorological conditions (i.e. global radiation (R<sub>g</sub>) within 50 Wm<sup>-2</sup>, air temperature within 2.5 °C and vapour pressure deficit (VPD) within 5.0 hPa). Initially, the averaging window was 14 days (i.e. ± 7 days) but this was extended to 28 days if there were no valid data within the immediate 14 day window. If temperature and VPD data was missing as well as NEE, the same approach was used (only using R<sub>g</sub>), but the averaging window was not extended beyond 14 days. If all meteorological variables were also missing, NEE values were replaced using the mean diurnal variation technique, starting with a window size of one day, and incrementally extending the window size until all gaps were filled. In this study the majority of gaps in NEE data were filled with the average of data from within a 14 day window.

### b) Flux partitioning

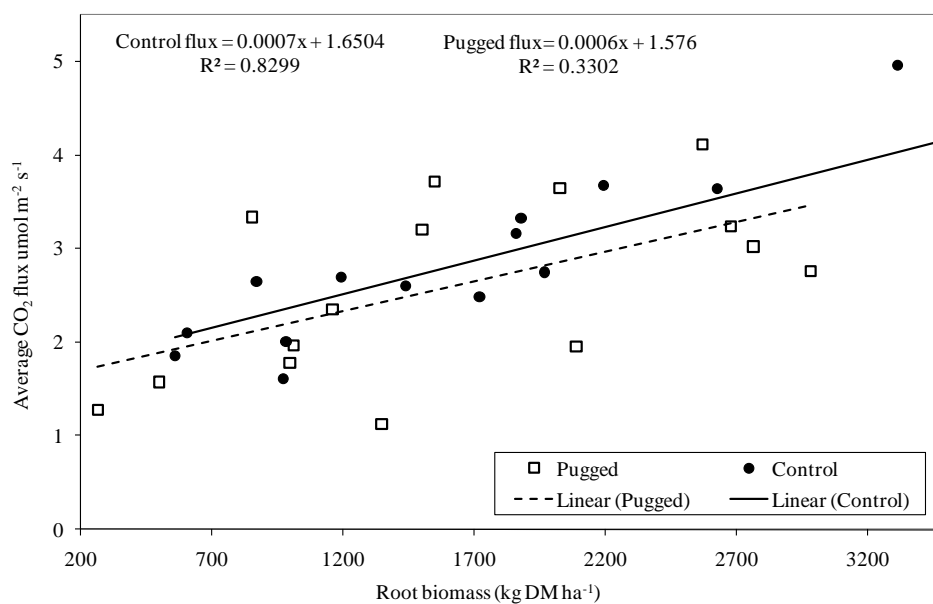
The flux partitioning model is based around the Lloyd and Taylor (1994) regression model. The dataset was divided into day and night based on a global radiation threshold (night <20 Wm<sup>-2</sup>). The night-time dataset was then split into

consecutive 10 day windows and the Lloyd and Taylor (1994) regression model fitted to NEE and air temperature data for each 10 day period (night-time NEE is equivalent to TER). Model parameters derived from the night-time data were then used in combination with daytime air temperatures to predict daytime TER. Gross primary production (GPP) was calculated by subtracting NEE from TER.

## APPENDIX C: RELATIONSHIP BETWEEN PASTURE ACCUMULATION AND CO<sub>2</sub> EMISSIONS AND ROOT BIOMASS AND CO<sub>2</sub> EMISSIONS



C1. Relationship between total pasture accumulation in individual survey chamber collars during the 41 day pugging trial and average CO<sub>2</sub> flux from the same collars.



C2. Relationship between total root biomass (0 – 75 mm depth) at the end of the pugging trial and average CO<sub>2</sub> flux from individual collars over the duration of the 41 day trial.





## **APPENDIX D: CD-ROM WITH FURTHER INFORMATION**

The attached CD-ROM contains further information relevant to this research. The content and layout of the CD is described in the file “Content and layout”.

Included is:

- Raw data
- Matlab scripts and functions used for data analysis and to generate figures
- Photographs of the EC field site and cultivation and pugging trials