Carbon dynamics of a dairy pasture: annual balance and impact of cultivation

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ABSTRACT

Maintenance of soil carbon (C) content is important because a relatively small percentage change in the global soil C store has the potential to cause a large change in atmospheric CO_2 concentration. Losses of soil C can also lead to a decline in soil quality and its capacity to be productive and carry out other services such as the filtering of pollutants. Globally, research on soil C dynamics has largely focused on forests, croplands and natural grasslands, while intensively grazed pasture has received much less attention. In New Zealand, the dynamics of soil C content and C cycling in intensively grazed dairy systems are poorly understood but large losses of soil C (1 t C ha^{-1} yr^{-1}) have recently been reported for grazed dairy pastures.

The objective of this research was to build on current knowledge of the C balance of intensively grazed dairy farm systems. To achieve this objective, net ecosystem CO_2 exchange (NEE) and water use efficiency (WUE) were measured over intensively grazed dairy pasture using eddy covariance from 15 December 2007 to 15 December 2009. Net ecosystem carbon balances (NECB) were then calculated for 2008 & 2009 from NEE measurements combined with measurements and estimates of C imports (feed) and C exports (milk, silage, methane). A further objective was to determine the impact of periodic cultivation of contrasting soils on the C balance of a dairy farm. To achieve this objective, measurements of soil CO_2 emissions were made using the closed chamber technique following the cultivation of three paddocks of Horotiu soil (Typic Orthic Allophanic) and three paddocks of Te Kowhai soil (Typic Orthic Gley).

Annual NEE of the farm was -1,212 ± 500 kg C ha^{-1} for 2008 and -2,280 ± 500 kg C ha^{-1} for 2009. Including imports and exports of C to the farm resulted in an annual NECB of -199 ± 500 kg C ha^{-1} and -1,014 ± 500 kg C ha^{-1} for 2008 and 2009, respectively. Applied uncertainty is at 90% confidence bound and derived from previous studies reported in the literature. The site was a net sink of C during both 2008 and 2009 in agreement with EC studies performed over grasslands in Europe. The large difference in NEE and NECB between years was due to a drought in 2008, when the site was a C source for the first four months of this year. Average daily water use efficiency (WUE) for 2008 (4 g C kg^{-1} H_2O) and 2009 (4.2 g C kg^{-1} H_2O) were not substantially different between years and agreed with international field and laboratory studies for pasture.

Soil CO_2 loss following cultivation was measured using the closed chamber technique. During the period of cultivation photosynthesis ceased, and potential C input (NEE) to pasture during this time was estimated at -750 kg C ha^{-1} from the adjacent EC study site. To calculate the maximum net soil CO_2-C loss the potential C input from photosynthesis (NEE) must be added to measured CO_2 emissions. Total soil C loss from the Te Kowhai was between 2,880 kg C ha^{-1} (CO_2 flux only) and 3,742 kg C ha^{-1} (CO_2 flux + NEE) while the Horotiu soil lost between 2,082 kg C ha^{-1} (CO_2 flux only) and 2,944 kg C ha^{-1} (CO_2 flux + NEE). The significant difference in C loss between the two soils was likely a result of their contrasting clay mineralogy and drainage. The Horotiu soil contains allophanic clays with a very high specific

surface area, which protects soil C from decomposition. Additionally, poorly drained soils such as the Te Kowhai tend to lose more C following cultivation due to aeration caused by cultivation which increases oxygen penetration into the soil and accelerates decomposition of soil C.

Based on these results this grazed pasture was a net sink of C for 2008 and 2009 which is in contrast to the measured decline of 1 t C ha^{-1} yr^{-1} from New Zealand's flat to rolling dairy pastures. Cultivation of dairy pasture soil resulted in net C losses, however, these losses were not large enough to account for the measured decline in soil C from New Zealand's flat to rolling dairy pastures. Further research is required to investigate long term soil C recovery following initial cultivation of pasture in order to be confident of this conclusion.

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3. ANNUAL CARBON BALANCE OF DAIRY PASTURE

4. IMPACT OF CULTIVATION ON SOIL C

ABBREVIATIONS AND SYMBOLS

AR	Autotrophic respiration	
C	Carbon	
CO_2	Carbon dioxide	
CH_4	Methane	
DM	Dry matter	(kg DM ha^{-1} yr^{-1} or kg C ha^{-1} yr^{-1})
E	Evaporation	(kg H_2O m^{-2} d^{-1})
EC	Eddy covariance	
GPP	Gross primary production	(kg C ha^{-1} yr^{-1} or g C m^{-2} d^{-1})
H	Sensible heat flux	(W m^{-2})
HR	Heterotrophic respiration	
λE	Latent heat flux density	(W m^{-2})
NECB	Net ecosystem carbon balance	(kg C ha^{-1} yr^{-1})
NEE	Net ecosystem exchange	(kg C ha^{-1} yr^{-1})
NEP	Net ecosystem production	(kg C ha^{-1} yr^{-1})
PPFD	Photosynthetic photon flux density	(µmol m^{-2} s^{-1} or mol m^{-2} d^{-1})
Rg	Global radiation	(MJ m^{-2} d^{-1})
TER	Total ecosystem respiration	(kg C ha^{-1} yr^{-1})
θ_v	Volumetric soil moisture content	(% v/v)
u_*	Frictional velocity	(m s^{-1})
VPD	Vapour pressure deficit	(kPa)
WUE	Water use efficiency	(g C kg^{-1} H_2O)

CHAPTER ONE

Introduction

1.1 Background

Soils are an important part of the global carbon (C) cycle as they contain the largest terrestrial pool of actively cycled carbon. The mass of C stored in the first metre of soil (2000 Pg C) is greater than the mass stored in above ground biomass (500 Pg C) and in the atmosphere (760 Pg C) combined (Janzen 2004). As a consequence, a small change to this C store has the potential to significantly modify the carbon dioxide (CO_2) concentration of the atmosphere (Schlesinger & Andrews 2000; Amundson 2001). Change in the soil C store also has a significant role in the chemical, biological and physical properties of soil. Soil C increases soil charge characteristics, cation exchange and buffering capacities as well as providing a source of energy and nutrients for plants and soil biota. Increasing soil C content also improves soil aggregate stability, resistance and resilience to compaction, and hydraulic properties (Haynes 2005).

The amount of C stored in soil varies depending on soil type, climate and the way land is used. If a particular land use does not change for a long period of time soil C content will eventually become stable, this is termed “steady state”. Steady state soil C occurs when the input of C derived from the fixation of CO_2 via photosynthesis equals the output of C as CO_2 from ecosystem respiration (Haynes 2005). This steady state can be disrupted by changes to annual input of C from above and below ground sources or by changes in decomposition of soil C. Changing land use or land management practices has frequently been shown to alter both inputs and outputs of C (Guo & Gifford 2002). For example, conversion of pasture to continuous cropping results in a net loss of soil C due to accelerated decomposition and a reduction of organic matter inputs, while converting cropped land to pasture, secondary forest or plantation forest results in a net gain of soil C (Guo & Gifford 2002). Conversion of forest to continuous cropping results in a net loss of soil C (Guo & Gifford 2002), while converting forest to pasture causes no significant change in soil C (Murty *et al.*

2002), however, in New Zealand the conversion of pine forest to pasture has resulted in losses of soil C (Tate *et al.* 2005).

Globally, 26% of ice free terrestrial surface area is under grazing (Steinfeld *et al.* 2006), and as this land use continues to intensify it is important that we build knowledge surrounding inputs and outputs of C to these systems. A better understanding of carbon budgets for intensively grazed pasture is important for New Zealand agriculture. New Zealand's dairy industry is intensifying with milk solid production per hectare rising by 34% from 1994 to 2002 (Parliamentary Commissioner for the Environment 2004). Intensification has increased periodic cultivation frequency on intensively grazed dairy systems from once every 10 – 15 years to once every 5 – 10 years (Pasture Renewal Charitable Trust 2009).

Early research into soil C in New Zealand determined that the conversion of land to improved pasture would result in a new soil C equilibrium within 7 – 41 years depending on soil type (Jackman 1964). Schipper & Sparling (in press) compared 10 soils that had been converted from scrub to pasture using data derived from work originally performed by Jackman (1964). They found that when the full 0 – 30 cm depth was considered, two soils showed significant increases in total C, two soils showed significant declines in total C and the remainder of the sampled soils showed no significant change in total C. However, on average soil was accumulating C through the 0 – 30 cm depth over 0 – 5 years, 5 – 25 years and 25 – 50 years (Schipper & Sparling in press).

Tate *et al.* (1997) re-sampled 43 long-term pasture soils that had been initially sampled 30 – 50 years previously for soil C and found that there had been no systematic temporal changes in soil C for mineral soils over time and suggested that soil C in NZ pastures was at steady state. However, the majority of soils sampled by Tate *et al.* (1997) were not as intensively managed as more recent pasture management. In contrast, Schipper *et al.* (2007) identified that New Zealand soils under pasture on flat to rolling land (commonly under intensive dairy land use) had lost about 1 t C ha^{-1} yr^{-1} for the last 17 – 30 years. Losses on a similar scale have also been measured internationally. Meersmans *et al.* (2009)sampled total C within the top metre of Belgium's agricultural soils and identified a loss of 0.6 t C ha^{-1} yr^{-1} from poorly drained agricultural soils over a 40 year period. Bellamy *et al.* (2005) reported losses of a similar magnitude for soils in England and Wales over a period of 25 years, however, not all of these soils were under pasture. The reason for the large changes in C of pasture soil is not clear and there is a need to develop annual C balances to determine the effect of intensive land management on soil C stocks on an annual basis.

Changes in soil C are commonly identified by directly sampling soils from locations that had been previously sampled, the change in soil C content over time gives an indication of whether soil is gaining or losing C (Schipper *et al.* 2007; Meersmans *et al.* 2009). This approach is rather insensitive to small changes in soil C and cannot be applied to the measurement of C changes on an annual basis. There is a need to establish ecosystem C balances on an annual scale as this will improve understanding of how different management practices alter changes in soil C stock. The eddy covariance (EC) measurement technique allows net ecosystem CO_2 exchange (NEE) to be measured (Baldocchi 2003). Estimates and measurements of secondary C fluxes from imports such as feed and exports such as product, erosion, leaching and methane loss can then be combined with NEE to create net ecosystem C balances (NECB). Research into the components of annual NECB in New Zealand's intensively grazed pasture is currently expanding. The first annual NECB for intensive pasture constructed using eddy covariance in New Zealand was for a dairy pasture on an organic soil (Nieveen *et al.* 2005), recently an annual C budget was established for a dairy pasture on a mineral soil (Mudge 2009). The study by Mudge (2009) was for a single year which experienced a severe drought. Annual C balances for intensively grazed pasture on mineral soil during a range of climatic conditions are needed to enhance understanding of soil C dynamics.

There are many pasture management practices that alter C inputs and outputs which need to be quantified; on such practice is the periodic cultivation of intensively grazed pasture for pasture renewal and crop production. Cultivation of soil is well documented to break up aggregates and expose previously protected C to microbial decomposition (Six *et al.* 2004; Grandy & Robertson 2007), however, there is little research into the impacts of periodic cultivation (Conant *et al.* 2007). Parfitt (2009) determined that soils with a high allophane or imogolite content had higher contents of soil organic matter and were less likely to lose soil C following cultivation. Comparative studies of the soil C loss from allophanic and non-allophanic soils under long term (>20 years) continuous cultivation have been performed (Parfitt *et al.* 1997; Parfitt *et al.* 2002), however, there are no available studies that compare CO_2-C losses between allophanic and non-allophanic soils following a single cultivation event on an intensive dairy pasture.

Eddy covariance can also be applied to measure ecosystem water use efficiency (WUE) which is the ratio between C assimilation and evaporation (*E*) which provides an indication of the mass of water required to fix a mass unit of C (Aires *et al.* 2008). The determination of ecosystem WUE can ultimately enhance our ability to predict how climate change can affect the C and energy budgets of various ecosystems, one such ecosystem that is not well understood is grazed pasture. WUE at the individual leaf scale is well understood, however, large scale studies have primarily focused on determining WUE of forest and cropland (Tong *et al.* 2009). Research into ecosystem scale WUE of a typical New Zealand dairy pasture would provide increased knowledge on the controls of photosynthetic input of C to intensively grazed soils. Analysis of WUE during drought (2008) and a cold winter (2009) will be beneficial as there are currently few studies which have analysed inter-annual variation in WUE from climatically contrasting years.

1.2 Aims and Objectives

The aim of this research is to build on current knowledge of the C dynamics of intensively grazed dairy farm systems and quantify the impact of periodic cultivation on an annual C balance.

To achieve this, the following objectives were set;

1. Examine inter-annual variability in NEE and WUE over 2 contrasting years.
2. Produce a net ecosystem C balance for 2009.
3. Determine the CO_2-C loss associated with a single spring cultivation event of adjacent allophanic and non-allophanic soils and evaluate the likely impact on the NECB.

1.3 Outline of thesis

Chapter 2 is a review of current and historic literature relating to soil C and its measurement with a specific focus on the soil C balance of New Zealand's intensively grazed pasture. The impact of cultivation on soil C is also reviewed along with a summary of factors that control respiration from soils. Chapter 3 presents an annual NECB for the study site and the variability of net ecosystem exchange and water use efficiency from the site over a two year period which includes a 100 year drought. Chapter 4 presents results from a study of the impact of a one-off cultivation event on two contrasting soils. Chapter 5 summarises the findings of this thesis and presents recommendations for future research.

CHAPTER TWO

Literature review

2.1 Purpose of literature review

This literature review summarises research relating to soil carbon (C) cycling with a specific focus on the changes to this cycle following cultivation of pasture and cropped land. The first section describes the nature of soil C, followed by a discussion on the terminology used in typical C balances of intensively grazed pasture systems. The methods used to measure changes in soil C are then described, followed by a description of the inputs and outputs of C on a typical New Zealand dairy farm. Previous C balances established for grassland ecosystems in New Zealand and other countries are summarised. An overview of the impact of periodic cultivation on soil CO_2-C flux is presented including a discussion on the impact of soil moisture and temperature on soil CO_2-C flux. Previous research into ecosystem water use efficiency is also presented and the final section of the review identifies major gaps in current knowledge that will be addressed in this thesis.

2.2 Soil Carbon

Soil C makes up about 58% of soil organic matter (SOM) (Post *et al.* 2001) with the remainder of SOM composed of hydrogen, sulphur, nitrogen and phosphorous (Johnston *et al.* 2009). SOM is mainly comprised of humus which is a blend of plant and animal residues and by-products at various stages of decomposition (Lal 2004). Chemically, humus ranges from compounds that are easily decomposed by microorganisms to compounds that are only slowly decomposable. The location of organic matter also plays a role in decomposition, SOM located on the exterior of soil aggregates is more easily decomposed than SOM protected in the interior of an aggregate (Amundson 2001).

SOM can be separated into light and heavy fractions (Christensen 2001). The light fraction is composed of free particulate plant and animal residues that are not bound to mineral matter. Light fraction SOM accumulation can be large in permanently vegetated soils such as those under grassland (Post *et al.* 2001). The turnover rate of the light fraction is relatively rapid and ranges from months to years, however, this rate can be dependent on land use (Post *et al.* 2001). If soil is cultivated, C within light fraction SOM is rapidly decomposed because it has little physical protection from decomposition (Beare *et al.* 1994; Biederbeck *et al.* 1994).

The heavy fraction of SOM constitutes the bulk of soil C and is largely physically protected from microbial decomposition. The heavy fraction is stabilized in organo-mineral complexes with clay and silt particles which reduces access of this SOM pool to decomposers (Post *et al.* 2001). Due to this stabilization, the turnover time for heavy fraction SOM is in the order of decades while some particularly recalcitrant SOM has turnover times on the scale of thousands of years (Jenkinson 1990). Six *et al.* (2004) describes organic matter dynamics at the micro-aggregate scale in depth, the reader is referred to this paper for further information.

Differences in clay mineralogy result in different degrees of soil C protection (Parfitt *et al.* 2002). For example, allophane and imogolite are two clay minerals which can protect SOM and increase the residence time of soil C. Soil C content within allophanic soils is greater than that for other soil orders because of C stabilization that occurs through the interaction of C with allophane and Al ions (Parfitt 2009). For example, at high elevations in Costa Rica increasing allophane content has been positively related to increasing C concentration (Powers & Schlesinger 2002) while in New Zealand, soil C content has also been related to the high extractable pyrophosphate Aluminium (Al_{py}) content in allophanic soils rather than specific clay content (Percival *et al.* 2000).

2.3 Terminology associated with annual carbon balances

Soil C content ultimately derives from the balance of C inputs from photosynthesis and C losses from respiration. The difference between the output of C from total ecosystem respiration (TER) and the input of C from gross primary productivity

(GPP) or photosynthesis is referred to as net ecosystem exchange (NEE). The primary fluxes of C within an ecosystem are highlighted in Figure 2.1.

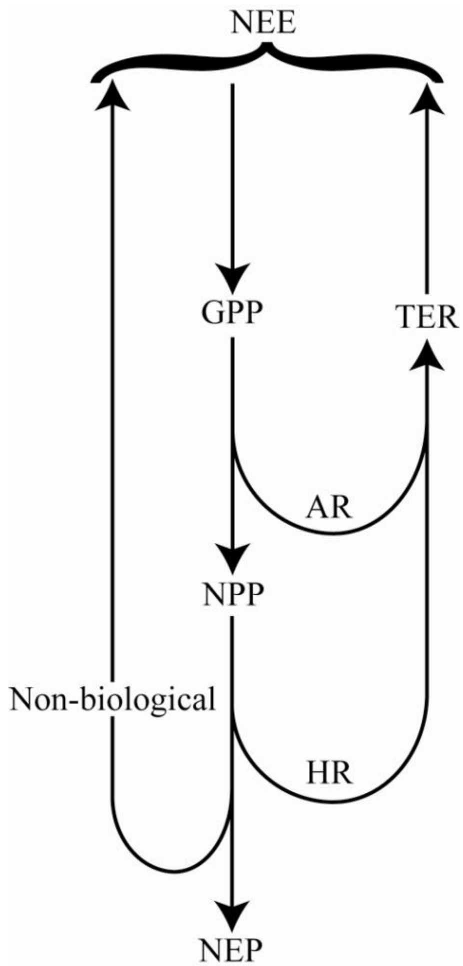

Figure 2.1 Transfers and flows of C between atmosphere and ecosystem. NEE = net ecosystem exchange, GPP = gross primary production, TER = total ecosystem respiration, NPP = net primary production, AR = autotrophic respiration, HR = heterotrophic respiration, NEP = net ecosystem production, Non-biological CO_2 flux is that derived from fires and the decomposition of SOM by light. A full description of these flows of C are given in text. Figure adapted from Luyssaert *et al.* (2007).

Total CO_2 entering the ecosystem via photosynthesis is referred to as gross primary production (GPP) while total CO_2 respired from the ecosystem's organisms (micro-organisms, plants, animals) is referred to as total ecosystem respiration (TER) (Luyssaert *et al.* 2007). Respired CO_2 can be derived from an autotrophic source or a heterotrophic source. Autotrophic respiration (AR) is the respiration from the above- and below-ground components of plants (Chapin *et al.* 2006). Heterotrophic respiration (HR) is the respiration sourced from soil fauna (e.g. microbes) and animals supported by the soil (e.g. cattle) (Hanson *et al.* 2000; Kuzyakov 2006). The partitioning of these two CO_2 sources is poorly understood, however, for ryegrass pasture it is generally assumed that autotrophic respiration contributes about 45% to the total ecosystem respiration (Hanson *et al.* 2000; Kuzyakov *et al.* 2001).

Net primary production (NPP) is defined as the amount of photosynthate not used for autotrophic respiration and is the difference between GPP and AR (Luyssaert *et al.* 2007). Most of the photosynthate not used for AR in an intensively grazed ecosystem is used to produce foliage and root biomass, a smaller amount of NPP is lost as volatile organic compounds and methane, exuded through roots and transferred to mychorrhizal fungi (Luyssaert *et al.* 2007).

Net ecosystem production (NEP) is defined as the difference between GPP and TER (Chapin *et al.* 2006). Net ecosystem exchange (NEE) is defined as the net exchange rate of CO_2 between the land surface and the atmosphere (Baldocchi 2003). NEE differs from NEP because NEE includes non-biologically sourced CO_2 fluxes such as those from fires and the decomposition of organic matter by light (Chapin *et al.* 2006; Rutledge *et al.* in press). NEE is generally used by atmospheric scientists, so inputs of C to ecosystems are defined as negative as they represent a loss of CO_2 from the atmosphere. NEP is used by ecologists and is opposite in sign to NEE (Chapin *et al.* 2006), a positive NEP represents C stored either as increased biomass (plant & animals) or increased soil C while negative NEP either represents a loss of biomass or soil C.

2.3.1 Net ecosystem carbon balance

To compile an annual C balance for an ecosystem, all C inputs and outputs must be accounted for. Grazed pastoral ecosystems can also gain C through import of feed and manure as fertiliser and lose C due to harvesting crops, exporting product (meat, milk, and plant biomass), methane emission, and leaching and erosion of C. To account for these sources of C, Chapin *et al.* (2006) recommended the introduction of a new term; net ecosystem carbon balance (NECB) to represent the overall ecosystem C balance from all physical, biological and anthropogenic sources and sinks within the ecosystem.

Equation 2.1 has been developed from equations presented by Soussana *et al.* (2007) and Chapin *et al.* (2006) to calculate the NECB of an intensively grazed dairy farm.

$$NECB = NEE + (-F_{import}) + F_{harvest} + F_{CH4} + F_{product} + F_{leach} + F_{erosion} \qquad \text{Equation 2.1}$$

Where, F_{import} is the C brought into the ecosystem as feed or manure, $F_{harvest}$ is the C exported from harvested plant biomass, $F_{product}$ is the C exported from the ecosystem as milk, meat or wool, F_{CH4} is the C lost via methane (CH_4) emissions from grazing ruminants and manure, F_{leach} is C lost via leaching through soil to groundwater and $F_{erosion}$ is C lost through erosion.

2.4 Methods for assessing changes in soil carbon

The assessment of soil C change requires the application of appropriate techniques to measure the response of soil C to different land use and management practices. There are two main approaches for assessing C gain or loss from soil over short or long time periods; direct measurement of soil C content or measurement of CO_2 exchange using chamber techniques or eddy covariance combined with estimates of other C losses (leaching, methane, erosion and product) and C gains (feed imports) (e.g. Equation 2.1).

2.4.1 Direct measurement

The most common way to directly assess changes in soil C content is to take soil samples, measure the C concentration and bulk density and then calculate mass of C per area (kg C ha^{-1}) (Post *et al.* 2001). This technique has been applied by numerous studies (West & Post 2002), however, this approach can be flawed if soils are sampled to inadequate depth (Baker *et al.* 2007; Blanco-Canqui & Lal 2008). When re-sampling soil, changes in bulk density must also be taken into account to calculate changes in C with time, as bulk density can change with time especially when comparing the change in C content of a soil where land use has changed (e.g. pasture to cropping). To overcome this problem, soil samples can be taken to a greater depth which will be more stable (e.g. > 600 mm) or corrected to an equivalent mass so that fair comparisons can be made (Post *et al.* 2001).

Ensuring adequate sampling and correction procedures are in place is critical for estimating rates of change in soil C. Baker *et al.* (2007) showed that apparent gains in soil C for soils converted from conventional tillage to conservation tillage were likely due to inappropriate sampling protocol rather than actual soil C sequestration. This study found that soils sampled to 30 cm or less displayed changes in soil C content after long term cultivation, while soils sampled to greater depths did not display long term changes in soil C following conversion from conventional cultivation (mouldboard plough) to less intensive cultivation methods (no-till). Similarly, Blanco-Canqui & Lal (2008) showed that sampling depth was crucial to conclusions drawn from direct measurements of soil C. Their study found that when soil was sampled to 60 cm there was no difference in C content between conventional tillage and conservation tillage, however, if soil was only sampled to 10 or 30 cm depth, significant differences were found between the two land management practices. The reason that sampling to 60 cm produces no significant difference between conventional cultivation and conservative cultivation was due to the deep burial of crop residues and the deeper rooting depth that occurs when conventional cultivation was used (Blanco-Canqui & Lal 2008). Soil C in the top 30 cm of soil was significantly different between treatments because the topsoil is prone to rapid perturbations and decomposition by the increased near-surface microbial activity and high fluctuations in soil temperature and moisture regimes (Blanco-Canqui & Lal 2008).

Salome *et al.* (in press) performed a 51 day incubation of topsoil (0-10 cm) and subsoil (80-100 cm). This study found that, in spite of lower C contents, disturbing the structure of the subsoil resulted in a 75% increase in mineralization rate of organic matter, while performing the same treatment on the topsoil resulted in no significant change in mineralization rate. The study of Salome *et al.*(in press) has highlighted that soils cannot be considered to behave in a uniform way through the entire soil depth and it is important to consider the impact of structural disruption on subsoils.

Another limitation to the direct measurement of soil C is the length of time needed between samplings. Treatment effects generally mean that annual changes in soil C are small in comparison to total C stocks and significant time is needed before significant changes in soil C can be detected against the high spatial variability of soil C stocks (Ammann *et al.* 2007).

2.4.2 Chamber measurements

There are numerous chamber techniques used to measure CO_2 emission from soil; techniques include the alkali-trap, static chamber, closed chamber and dynamic chamber. Chamber techniques have been used to measure soil respiration since the 1920's and have developed with time (Rochette & Hutchinson 2005). Prior to 1985, the dominant chamber technique was the alkali trap method where a closed chamber which included a suspended vessel containing a known amount of dissolved alkali (NaOH or KOH) was placed over the soil surface to trap CO_2 for 12 – 24 hours (Rochette & Hutchinson 2005). The amount of CO_2 trapped over the deployment period was then determined via titration. Rochette *et al.* (1992) compared the alkali trap and dynamic chamber method (for description of dynamic chamber method see following page) under field conditions and found that the alkali trap consistently underestimated soil CO_2 flux when compared to the dynamic chamber. This was likely to be caused by one or more of the following; cooler soil temperature within the alkali trap chamber would reduce the rate of soil respiration, an increase in alkali trap chamber air CO_2 concentration would lower the CO_2 gradient between the soil and the chamber which would effectively reduce CO_2 fluxes, or the CO_2 absorption of the alkali solution may decrease over time which could also reduce CO_2 flux measurements (Rochette *et al.* 1992). This consistent underestimation has lead to the conclusion that results from previous studies which applied the alkali trap method are unreliable (Rochette & Hutchinson 2005). However, the alkali trap method is still an inexpensive option that can be applied to obtain an approximate measurement of soil respiration in remote locations where other chamber techniques may not be viable.

The static chamber method uses a closed chamber to determine the rate of CO_2 accumulation. The method involves removing gas samples from a sealed chamber via a sampling port at regular intervals over a known time period (Grandy & Robertson 2006). The collected gas samples are then transferred to a laboratory where CO_2 is measured by either a gas chromatograph (GC) or an infrared gas analyser (IRGA). Collecting samples in the field and processing them in the laboratory is labour intensive which increases the cost and time associated with sampling. Portable infrared gas analysers have been developed which allow processing time to be reduced as all calculations are done while chamber measurements are being taken allowing data to simply be downloaded once sampling has been completed.

There are two common approaches that utilise a portable IRGA, these are; the closed chamber method and the dynamic chamber method. The closed chamber method uses a chamber which seals itself over a protruding soil collar, once the closed chamber forms a seal with the collar the CO_2 concentration within the chamber increases. The air within the chamber is then pumped in a loop between the chamber itself and the portable IRGA allowing CO_2 concentration readings to be made quickly and accurately (Luo & Zhou 2006).

The dynamic chamber method is similar to the closed chamber method, however, this method avoids an increase in chamber CO_2 concentration by passing air from outside the chamber through the chamber at a constant rate. CO_2 concentration of air entering and leaving the chamber are both measured, these values are combined with chamber air temperature and pressure to calculate soil respiration (Rochette & Hutchinson 2005).

Chamber methods remain popular as they allow soil respiration to be measured easily at a small scale. The advent of the portable IRGA and chamber has allowed sampling and processing time to be reduced, however, there are still the same inherent problems with the chamber method. Measurements taken at a small scale must be repeated at an adequate number of locations to ensure spatial variation is taken into account, and these small scale measurements must then be extrapolated to scales far larger than the size of the soil collar from which they were measured. All of the chamber methods discussed disturb natural soil respiration conditions although some cause more disturbance than others (e.g. dynamic chamber is superior to the alkali trap method) (Rochette & Hutchinson 2005).

2.4.3 Eddy covariance

The eddy covariance technique (EC) determines the rate of CO_2 exchange across the interface between the atmosphere and plant canopy by measuring the covariance between fluctuations in the vertical wind speed and CO_2 mixing ratio at high frequencies (20 Hz) from the near surface atmosphere (Baldocchi 2003). These measurements are taken using a sonic anemometer to measure three dimensional wind-speed and either an open path or closed path infrared gas analyser to measure CO_2 flux. It is important to set the eddy covariance instruments at a height which allows adequate fetch for the study area. Fetch is the distance over which measurements are made across the study area, and is approximately 100 times the height at which the eddy covariance instruments are mounted, therefore, the higher the instruments, the larger the study area over which the measurements are made (Baldocchi 2003). The EC technique is most accurate when used on flat sites with steady atmospheric conditions and homogenous vegetation (Baldocchi 2003). This list of site prerequisites can be met by intensively grazed dairy farms in New Zealand, which are often situated on flat terrain with uniform vegetation in each paddock. However, even with an ideal site there are still potential errors associated with measuring CO_2 exchange using EC (Baldocchi 2003).

Measurements of night-time CO_2 flux can be underestimated due to a combination of insufficient turbulent mixing, incorrect measurement of CO_2 in the air space and soil, and the drainage of CO_2 out of the canopy at night (Baldocchi 2003). This underestimation of night-time fluxes is important because night-time NEE is used to estimate day-time TER. At night, GPP is zero because there is no light to drive photosynthesis, which allows the development of a site-specific relationship between night-time NEE and soil or air temperature which can be used to estimate half hourly TER during the day when day-time soil or air temperature is known (Luyssaert *et al.* 2009). GPP can then be estimated by subtracting day-time TER from NEE (Reichstein *et al.* 2005).

Applying night-time NEE to estimate day-time TER may be problematic because under similar conditions, day-time foliar respiration is consistently less than at night-time (Luyssaert *et al.* 2009). Day-time foliar respiration is reduced due to the impact of photorespiration occurring in the light (Atkin *et al.* 1998) and by re-fixation of respired CO_2 (Luyssaert *et al.* 2009). The difference between day-time and night-time TER makes the use of night-time NEE measurements to estimate day-time TER problematic as an overestimate is likely to occur (Luyssaert *et al.* 2009). This overestimation can be reduced by applying gap filling models which use variable parameter values to fill gaps in EC data such as that presented by Reichstein *et al.* (2005).

Analysis of surface energy balance closure at EC sites suggests that turbulent fluxes at some sites are 10-30% too small to close the energy balance which suggests that CO_2 fluxes are also underestimated (Baldocchi 2003). The discrepancy can be corrected by adjusting CO_2 flux densities in proportion to the underestimated energy balance closure, however, this approach involves reliance measurements of global radiation and soil heat flux from a small footprint directly beneath the tower, whereas eddy covariance measurements are sourced from a footprint with an are in the order of hundreds to thousands of square meters (Baldocchi 2003).

Due to the numerous sources of error associated with EC data (low wind speeds, ice or dew forming on instruments, data logger malfunction etc), gaps within the data occur frequently. On an annual scale, about 65% of data collected are usually retained, with the remainder being filtered out due to instrument error or low wind speed (Falge *et al.* 2001). To work with EC data on daily, monthly or annual time scales, a continuous data set must be derived by filling these gaps. There are numerous approaches to gap filling and the results produced (NEE, TER, and GPP) can be substantially different when different gap filling approaches are applied to the same filtered data set (Falge *et al.* 2001). Applying a gap filling method that has good overall performance such as that presented by Reichstein *et al.* (2005) can help to reduce this source of error (Moffat *et al.* 2007).

Eddy covariance has been applied globally to measure CO_2 exchange over natural and cut and carry grassland ecosystems, however, few studies have investigated intensively grazed pastures (Table 2.1). In New Zealand, C balances have been compiled for two intensively grazed dairy farms based on the eddy covariance measurements of NEE. The first annual C balance was for an organic soil (Nieveen *et al.* 2005), and recently an annual C balance was established for farm with mineral soils that experienced a drought (Mudge 2009).

2.5 Net ecosystem carbon balance

This section will cover the various inputs and outputs which must be accounted for in a net ecosystem carbon balance (NECB).

2.5.1 Inputs

The two main inputs of C to the net ecosystem C balance (NECB) on a New Zealand dairy farm are sourced from the fixation of CO_2 via photosynthesis and feed imported for stock.

2.5.1.1 Photosynthesis

The largest input of C into an intensively grazed ecosystem is plant fixation of CO_2 through photosynthesis (Bryne & Kiely 2006). Plants convert CO_2 from the atmosphere into organic C compounds through photosynthesis, these compounds are either used to grow plant tissue or as an energy source for the plant (Luo & Zhou 2006). Carbon fixed via photosynthesis can be transferred to the soil through the death of above ground biomass, and the consumption of pasture by livestock and subsequent return of the C through excreta. A major pathway of fixed C from the atmosphere to the subsoil is through the transfer of C to roots where C can be released via root exudates and root death (Saggar & Hedley 2001). The input of C to the soil is dependent on the amount of pasture available to fix CO_2 via photosynthesis. Based on this simple relationship, the current study has made an attempt to estimate the input of C derived from photosynthesis through the use of average above ground dry matter production.

Above ground dry matter (DM) production is commonly assessed on New Zealand farms to assist with management decisions. DM production is assessed to ground level as the amount of dry matter per hectare area (kg DM ha^{-1}) and various methods are applied to measure this (e.g. calibrated visual assessment, cage cuts and rising plate meters) (AgResearch 2002). Average above ground DM production from New Zealand's intensive pasture based dairy farms is about 15,000 kg DM ha^{-1} yr^{-1} (Clark *et al.* 2007) of which about 40% is C (Saggar & Hedley 2001), therefore, annual above ground DM production is equivalent to an input of about 6,000 kg C ha^{-1} yr^{-1} (Clark *et al.* 2007). Root production on an intensive dairy farm is almost equivalent to above ground production (Saggar & Hedley 2001). Therefore, total DM production (also termed net primary production, NPP) would be about 30,000 kg DM ha^{-1} yr^{-1} or 12,000 kg C ha^{-1} yr^{-1}. NPP is the amount of photosynthate not used for heterotrophic respiration which is calculated by subtracting autotrophic respiration (AR) from gross primary production (GPP) (Luyssaert *et al.* 2007).

To estimate average GPP for an intensive dairy farm using dry matter production it must be assumed that the system has reached steady state soil C content so that GPP is equal to TER (i.e. NEP = 0). This is a large assumption as this does not always occur in well watered pastures (see Table 2.1). TER can be calculated as NPP (above and below ground dry matter production) plus autotrophic respiration (AR) which is approximately 45% of TER under perennial ryegrass (Kuzyakov *et al.* 2001) as follows:

Assume GPP = TER,

TER = NPP + AR (assuming that all NPP is respired) Equation 2.3

Estimating GPP from total DM production requires TER to be partitioned into AR and HR, if the assumption is made that AR contributes 45% of respiration to TER (Kuzyakov *et al.* 2001) then Equation 2.4 can be applied to estimate GPP when DM production is known.

Given:

DM above and below ground = Total NPP = 12,000 kg C ha^{-1} yr^{-1}, TER = AR + HR and

AR = 0.45TER

Then TER = 12,000 kg C ha^{-1} yr^{-1} + 0.45TER,

TER = 21,818 kg C ha^{-1} yr^{-1} Equation 2.4

As steady state is assumed (see Equation 2.3) then GPP = 21,818 kg C ha^{-1} yr^{-1}

Figure 2.2 displays an estimate of the primary fluxes of C derived from Equation 2.4, for a pasture with dry matter production of 15 t DM ha^{-1} yr^{-1}. Although GPP has been estimated and there are a number of simplifying assumptions made, the values compare well with measured studies (Table 2.1). The main uncertainty with Equation 2.3 is the assumption the GPP = TER, to reduce this uncertainty GPP can be measured directly.

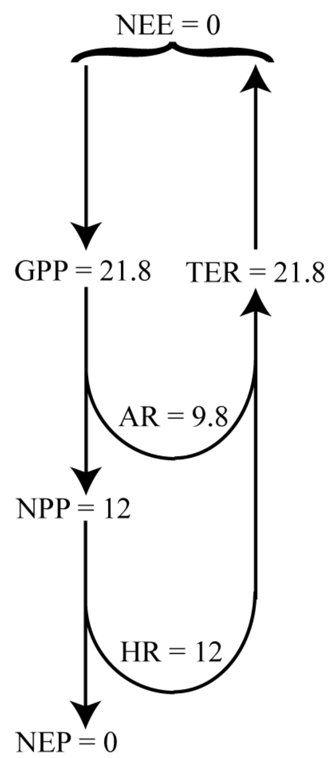

Figure 2.2 Estimate of C cycle for a New Zealand dairy farm producing 15 t DM ha^{-1} yr^{-1} of above ground dry matter and is at steady state. NPP is an estimate of the amount of C produced as above and belowground biomass. Units are tonnes of C per hectare per year (t C ha^{-1} yr^{-1}).

GPP can be derived using eddy EC measurements of NEE. Partitioning NEE into TER and GPP generally involves applying a model to night-time NEE and day-time air temperature to calculate TER, GPP is then calculated by subtracting TER from NEE (Reichstein *et al.* 2005). In New Zealand, Nieveen *et al.* (2005) applied this technique and measured a GPP of 13,486 kg C ha^{-1} yr^{-1} from a grazed dairy pasture (perennial rye grass and white clover) on an organic soil. Similarly, Mudge (2009) measured a GPP of 19,488 kg C ha^{-1} yr^{-1} for an intensive dairy pasture (perennial rye grass) on a mineral soil during a dry year. Numerous other eddy covariance studies have measured GPP over grassland, with values ranging from 21,590 kg C ha^{-1} yr^{-1} for intensive permanent pasture grown specifically for silage in France (Ammann *et al.* 2007) to 5,240 kg C ha^{-1} yr^{-1} from grazed grassland in Portugal (Aires *et al.* 2008) (Table 2.1).

Further work is needed to obtain an accurate understanding of the magnitude of annual C fluxes in New Zealand pasture systems. This could be achieved by carrying out more work on the partitioning of fixed C in pasture systems and also making more direct measurements of C exchange using eddy covariance. If adequate measurements could be obtained, it would be useful to verify the calculations based on DM production made in this section with measurements made with EC.

2.5.1.2 Feed imports

In New Zealand, between 2003 and 2004, 40% of dairy farms imported some form of feed (Clark *et al.* 2007), 61% of farms imported hay and silage, 24% imported maize silage and 37% imported grain and meal feed (Clark *et al.* 2007). The dominant form of supplementary feed in New Zealand is hay or silage while the use of imported grain and meal feed such as palm kernel extract (PKE) is increasing (Clark *et al.* 2007). Although New Zealand’s dairy industry is pasture-based, supplementary feeding has benefits such as buffering fluctuations in pasture production, increasing milk solid production and improve body condition (Clark *et al.* 2007). When feed is not grown on farm, it is classed as an import which needs to be included in the C balance. Mudge (2009) stated that imported feed makes up approximately 10% of total feed consumed on a typical dairy farm and that annual pasture utilisation (pasture consumed by livestock) is about 80% of total above ground pasture production. This would suggest that total C input from imported feed on a dairy farm producing 6,000 kg C ha^{-1} yr^{-1} (of which 4,800 kg C ha^{-1} yr^{-1} is utilised) is about 480 kg C ha^{-1} yr^{-1}.

2.5.2 Outputs

There are numerous outputs of C which must be accounted for to develop a net ecosystem C balance (NECB) for a New Zealand dairy farm. The primary output is total ecosystem respiration (TER), however, other losses such as the export of product (milksolids), leaching, erosion and methane loss from livestock must all be included to complete a NECB.

2.5.2.1 Respiration

Total ecosystem respiration (TER) is the largest single source of C loss from intensively grazed pastures. Respiration can be classified as autotrophic or heterotrophic depending on the source of the CO_2 flux. Autotrophic respiration (AR) is derived from living plant tissue within soil and is also commonly referred to root/rhizosphere respiration (Hanson *et al.* 2000). Heterotrophic respiration (HR) is derived from the decomposition of soil organic matter by microbes and is also commonly referred to as soil respiration. The majority of studies that attempt to separate AR from HR have been performed on soils under forest or crop, however, estimates of AR contribution to TER have been made for pasture and range from 20% to 50% (Hanson *et al.* 2000; Kuzyakov 2002), but the contribution is generally concluded to be about 45% (Kuzyakov *et al.* 2001).

TER also includes respiration derived from livestock within the farm system. Jaksic *et al.* (2006) determined that each livestock unit (1 LU=1 550kg cow) respired approximately 70% of C consumed as food, with the remaining 30% of C converted into meat, milk, dung and urine. On an average dairy farm producing 15,000 kg aboveground DM ha^{-1} yr^{-1} (6,000 kg C ha^{-1} yr^{-1}) around 80% of available dry matter is consumed by live stock (Mudge 2009). This means 4,800 kg C ha^{-1} yr^{-1} is consumed by live stock and 70% of C consumed is respired (Jaksic *et al.* 2006). Therefore, from a typical New Zealand dairy farm around 3,360 kg C ha^{-1} yr^{-1} would be respired by livestock while the remaining 1,440 kg C ha^{-1} yr^{-1} would go into meat, milk, dung and urine production. This respiration from livestock contributes to total ecosystem respiration (TER).

2.5.2.2 Methane

Methane (CH_4) is responsible for approximately 38% of New Zealand's global warming potential (Waghorn & Woodward 2004). This is high on a global scale and is primarily driven by New Zealand's large scale pasture based agriculture. Agriculture in New Zealand is responsible for 88% of methane emissions of which dairy contributes 34% (Waghorn & Woodward 2004). Although these emissions appear large in terms of global warming potential on a mass basis the losses of C are minimal compared to the loss from TER.

Measured emissions of methane from stock during spring, mid-summer and late summer range from 0.247 kg CH_4-C LU^{-1} d^{-1} (Laubach & Kelliher 2004) to 0.259 kg CH_4-C LU^{-1} d^{-1} (Robertson & Waghorn 2002) to 0.296 kg CH_4-C LU^{-1} d^{-1} (Woodward *et al*. 2004). The average methane emission from the above studies is 0.267 kg CH_4-C LU^{-1} d^{-1} and the average stocking rate for a New Zealand dairy farm during the 2007-2008 season was 2.83 LU ha^{-1} which gives average annual CH_4-C loss of 276 kg C ha^{-1} yr^{-1}.

2.5.2.3 Product

The primary form of product leaving the dairy farm gate is milk. In New Zealand, the quantity of milk is defined as mass of milksolids (MS) and consists of milk fat and milk protein. Each kilogram of MS that leaves the farm is equivalent to 0.834 kg of C (Wells 2001). Average MS production for the 2006/2007 season for NZ was 934 kg MS ha^{-1} (Livestock Improvement Corporation. 2008) or 779 kg C ha^{-1} during an average year.

Traditionally, New Zealand dairy farms raise replacement heifers off-farm. This practice means that little C is lost in the form of meat, as heifers are reintroduced to the herd when their live weight is approximately the same as the older cows that leave the herd as culls. This method of replacement means that C loss in meat is negligible on a typical New Zealand dairy farm, assuming herd size is approximately in balance.

Growth and harvesting of silage and hay is common practice on most New Zealand dairy farms, however, the loss of C associated with this is minimal as the large majority of farms feed the stored silage or hay back to their animals when feed deficits occur. Some farms do harvest silage for export to other farms, exports of silage can be large during times of regional or national feed shortage such as the drought that occurred in the Waikato region during the summer period of 2007. Silage production on dairy farms is not intensive and would generally involve two harvests of 3,000 kg DM ha^{-1}, harvesting occurs across about 10% of the farm (E. Thom, pers comm., 2009[1]) so the contribution of C loss to the net C balance of a typical 120 ha dairy farm would be 240 kg C ha^{-1} yr^{-1} (6,000 kg DM at 40% C). However, as mentioned previously, this C would normally be fed to animals within the farm making C loss from silage export minimal.

2.5.2.4 Erosion

Hunt *et al.* (2004) estimated wind erosion from sandy bare soils in New Zealand's Mackenzie basin to contribute about 20 kg C ha^{-1} yr^{-1}. This loss is small compared to other losses and would be reduced even further from farms with good pasture cover where annual rainfall was greater. A recent study modelled erosion for New Zealand and found C loss from pasture ranged from 0 – 50 kg C ha^{-1} yr^{-1} while flat land under pasture experienced C loss due to erosion of less than 20 kg C ha^{-1} yr^{-1} (Scott *et al.* 2006). This reinforces that C loss due to erosion from an average flat dairy farm is negligible in the context of an annual C balance, although this loss is recognised as important at national scales from hill country.

[1] Personal communication with Dr. Errol Thom, Scientist, DairyNZ, Hamilton.

2.5.2.5 Leaching

Leaching of soil C is difficult to measure under field conditions, however, some lysimeter studies have successfully measured the amount of C leached from various soil types. Sparling *et al.* (2006) conducted a lysimeter study which measured the amount of C leached from four soil types with contrasting drainage. This study measured loss of 39.8 kg C ha^{-1} yr^{-1} from a pumice soil, 56.3 kg C ha^{-1} yr^{-1} from a gley soil, 36.5 kg C ha^{-1} yr^{-1} from an allophanic soil, and 241.8 kg C ha^{-1} yr^{-1} from a recent soil. The differences in leaching rate between soil types suggests that gley soils have greater losses of C due to leaching than allophanic soils but these losses are small compared to the leaching recorded for the recent soil. Similarly, Ghani *et al.* (2008) found that dissolved organic C (DOC) leaching was greatest from gley soils in a comparison of pumice, ash and gley soils. These studies demonstrate that leaching could contribute to total C output, particularly from poorly drained soils (Sparling *et al.* 2006; Ghani *et al.* 2008), however, this loss is still very small in comparison to the large losses of C associated with TER.

2.5.3 Estimated NECB for typical New Zealand dairy farm

An estimate of the approximate inputs and outputs of C on a typical New Zealand dairy farm is given in Figure 2.3. This figure has been compiled from values presented in section 2.5 and it is not intended to indicate net change in soil C but rather represent the size of the various inputs and outputs in relation to one another.

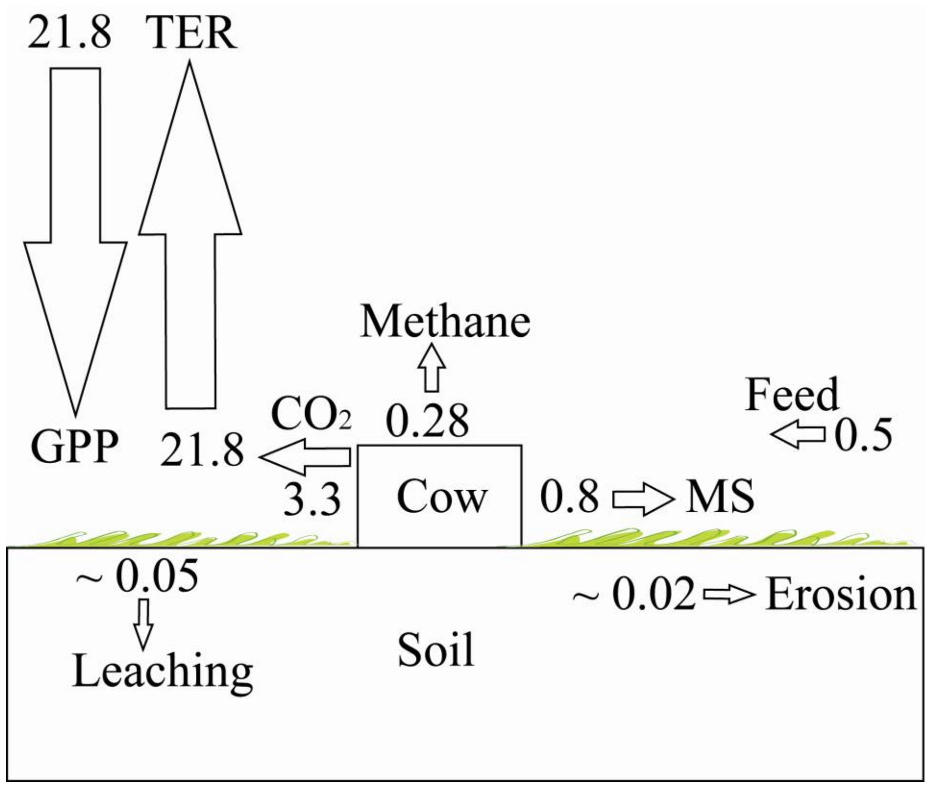

Figure 2.3 Schematic diagram of C fluxes from literature and estimates of GPP and TER for an average New Zealand dairy farm. All numbers are in tonnes of C per hectare per year (t C ha^{-1} yr^{-1}). TER is total ecosystem respiration, CO_2 is the contribution of livestock respiration to TER, GPP is gross primary production (photosynthesis), methane is the amount of C lost from livestock as methane, MS is the amount of C lost from the farm as milk solids (product). Feed is the average amount of C imported as feed from an external source, leaching and erosion are estimates of these losses as discussed in section 2.5.2.4 and 2.5.2.5. Export of plant biomass is not included as this figure is difficult to calculate for an average farm but is likely to be small (section 2.5.2.3).

2.6 Multi-year carbon balances for grazed pasture

Pasture based agriculture is important both in New Zealand and globally. Pastoral farmland dominates New Zealand's geography with over half of New Zealand's land area being defined as farmland and much of this area is under grazed pasture (Parliamentary Commissioner for the Environment 2004). Globally, 26% of the Earth's ice free terrestrial area surface is under grazed pasture (Steinfeld *et al.* 2006). While C balances based on estimated inputs (dry matter, feed) and outputs (methane, milksolids, leaching etc) can be achieved, this approach is too insensitive to determine whether pastures are a net sink or source of C. Other approaches are needed as creating annual C balances for this land use is of high importance due to the significant area that grazed pasture occupies, and the sensitivity of this ecosystem to change driven by changing management practice.

2.6.1 Measured soil C changes with time

Early research into soil C in New Zealand determined that the conversion of scrubland to improved pasture would result in a new soil C equilibrium within 7 – 41 years depending on soil type (Jackman 1964). Schipper & Sparling (in press) compared 10 soil chronosequences of scrub to pasture conversion, the data were derived from work originally performed by Jackman (1964). Schipper & Sparling (in press) found that when the full 0-30 cm depth was considered, two soils showed a significant increases in total C, two soils showed a significant decline in total C and the remainder of the sampled soils showed an increase but this was not significant. However, when averaged across all chronosequences, soils accumulated C through the 0-30 cm depth.

Tate *et al.* (1997) re-sampled 43 soils from 0 – 0.25 m and 0 – 1m depth under long term pasture that had been initially sampled 30-50 years previously for soil C and found that there had been no systematic temporal changes in soil C for mineral soils with time and suggested that New Zealand's soil C in NZ pastures was at steady state. In a subsequent study, Schipper *et al.* (2007) showed that New Zealand soils under pasture on flat to rolling land (commonly under dairy land use) had lost about 1 t of C per hectare annually for the last 17-30 years. Losses on a similar scale have also been measured internationally. Meersmans *et al.* (2009) sampled for total C within the top metre of Belgium's agricultural soils and measured a loss of 0.6 t C ha^{-1} yr^{-1} from poorly drained agricultural soils over a 40 year period. Bellamy *et al.* (2005) sampled to 0.15 m and reported losses of a similar magnitude for soils in England and Wales over a period of 25 years, however, not all of these soils were under pasture.

Following the identification of soil C loss for the last 17-30 years from flat to rolling pasture soils, Schipper *et al.* (in press) examined whether these losses were also occurring at a hill country site. This study showed that there were significant changes in the C & N content of hill country pasture but these trends were not unidirectional. Increases in soil C were measured for the first six years of the trial followed by either no change or declines in C over the next 17 years depending on slope class examined. The losses were attributed to a series of dry summers which decreased photosynthesis while microbial respiration continued (Schipper *et al.* in press).

While long-term losses of soil C in dairy pastures have been identified, it is unclear whether soil C stocks under intensive pasture are continuing to decline. This highlights the need to develop annual C balances of grazed pasture in order to determine the effect of intensive land management on soil C stocks.

2.6.2 Balances established using eddy covariance

Net ecosystem C balances (NECB) can be established on an annual scale using eddy covariance (EC) and physical measurements or estimates of other C imports and exports. One ecosystem that has the potential to be a major sink or source of C is agricultural pasture as these ecosystems are often intensively managed by farmers. Because these ecosystems are managed, they have the potential to fix CO_2 from the atmosphere through the adoption of management practices that reduce C loss and increase C gain. Globally, grassland accounts for about 32% of Earth's natural vegetation (Jaksic *et al.* 2006), and is now receiving renewed interest as a potential sink of atmospheric CO_2.

Table 2.1 summarises a range of previous eddy covariance studies that have investigated C balances over pasture. From these studies some generalisations can be made, pasture on mineral soils are generally a sink of C (Jaksic *et al.* 2006; Allard *et al.* 2007; Mudge 2009); whereas, managed pasture on organic soils are generally a source of C (Nieveen *et al.* 2005; Veenendaal *et al.* 2007). Grassland with high average annual temperatures and low rainfall generally results in NEE being close to zero (Xu & Baldocchi 2004). However, there can be considerable variation in NECB from year to year. Ammann *et al.* (2007) measured changes in NECB of 1,480 kg C ha^{-1} yr in 2003 and -40 kg C $ha^{-1}yr^{-1}$ in 2004 from an extensively managed field (over 30 grass, clover and herb species) in Switzerland from 2003 to 2004. In this case, the large loss of C from the system (1,480 kg C ha^{-1} yr) during 2003 was due to a summer drought. These environmental conditions produced a low NEE (-710 kg C ha^{-1} yr) and when product export was taken into account (2,190 kg C ha^{-1} yr), NECB became positive and therefore the ecosystem was a source of C. This compilation of current research highlights the need for more long-term assessments of NECB to be performed over grassland ecosystems during climatically contrasting years.

Soussana *et al.* (2007) calculated NECB values for nine contrasting grassland sites across a climatic gradient through Europe for a period of two years (Table 2.1). The study provides an indication of year to year variation in the NECB of a land use. Table 2.1 shows that all grassland sites were a sink of C with negative NEE values; however, variability between sites was high. There was also significant annual variability, for example Malga Arpaco, Italy experienced a 45% decline in C sink activity in year two of the study. Soussana *et al.* (2007) concluded that European grasslands are likely to act as large atmospheric sinks of CO_2, however, the number of study sites was relatively small and the authors recommended that more studies investigate CO_2 exchange from grassland sites using eddy covariance.

Table 2.1 NECB for various pasture systems measured using eddy covariance, all C balance values are in kg C ha^{-1} yr^{-1} unless stated otherwise (adapted from Mudge (2009)).

Reference	Soil type	Location	Vegetation	Management	Year	Mean annual temp (°C)	Rain (mm yr-1)	GPP	TER	NEE	Supplement & manure inputs	Supplement outputs	Animal product outputs	Methane from animals	Other	NECB
Aires *et al.* (2008)	Luvisol	Portugal	C3/C4 grassland	Grazed by sheep (Oct-Feb)	2004	14.7	364	5,240	5,730	490						
					2005	14.5	751	12,610	10,710	-1,900						
Allard *et al.* (2007)	Andosol	France	Permanent pasture	Continuous grazing (May-Oct)	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 the 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
Ammann *et al.* (2007)	Eutri-Stagnic Cambisol	Switzerland	Permanent pasture (7 species)	Intensive (cut for silage 4 times 200 kg N ha^{-1} yr^{-1})	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 species)	Extensive (cut for silage 3 times)	2002	9.6	1,479	17,140	13,620	-3,250		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
Brown *et al.* (2009)	Typic Orthic Gley	New Zealand	Perennial ryegrass and white clover	Grazed by cattle	2005*	11.7*	782*		19,500							
Byrne *et al.* (2007)	Gleysol with shallow peat layer	Ireland	Perennial rye grass and white clover	Dairy pasture grazed and cut for silage	2004	9.5	1,340			-2,900	-540		280	115	945[a]	-2,100
Hunt *et al.* (2004)	Typic Dystrustept	New Zealand	Tussock, *Heiracium*(43% bare ground)	Grazed by sheep and unfertilized	1998	9.9	446			90						
					1999	9.2	933			-410						

***averages not specific to year of study**

[a]775 kg from livestock respiration when in dairy shed and when housed for winter, also includes methane loss from manure spreading and dissolved organic C (DOC) leaching

Table 2.1 continued

Reference	Soil type	Location	Vegetation	Management	Year	Mean annual temp (°C)	Rain (mm yr-1)	GPP	TER	NEE	Supplement & manure inputs	Supplement outputs	Animal product outputs	Methane from animals	Other	NECB
Jaksic *et al.* (2006)	Surface water gley	Ireland	Perennial rye grass and white clover	Dairy pasture grazed and cut for silage	2002	10	1,785	16,730	14,800	-1,930			440	165	1,089[b]	-240
					2003	10	1,185	17,180	14,600	-2,580			440	165	1,089[b]	-890
Mudge (2009)	Typic Orthic Gley	New Zealand	Perennial rye grass	Grazing and cutting	2008	13.8	1,126	19,448	17,605	-1,843	-218	70	840	271		-880
Nieveen *et al.* (2005)	Peat	New Zealand	Perennial rye grass and white clover	Rotationally grazed dairy pasture	2002	13.8	1,281	13,486	13,531	45			738	278		1,061
Novick *et al.* (2004)		North Carolina	C4 grass pasture	Mowed annually in summer for hay	2001	15.5	1,145	12,020	12,990	970						
Veenendaal *et al.* (2007)	Clayey peat or peaty clay	The Netherlands	Perennial rye grass	Cut 3 times, grazed 2-3 times	2005	9.8*	793*			1,339	-1,540	4,100		350		4,230
Skinner (2008)	Typic Hapludault	Pennsylvania	Grass based pasture	Rotationally grazed by cattle and harvested for silage	2003-2006	9.7*	1,014*			270	-400	1,260	Not included			1,130
			Alfalfa based pasture	Rotationally grazed by cattle and harvested for silage	2003-2006	9.7*	1,014*			-650	-130	1,260	Not included			480

***averages not specific to year of study**

[b] 1,039 kg from livestock respiration when in dairy shed and when housed for winter, as well as 50 kg of DOC leaching

Table 2.1 continued

Reference	Soil type	Location	Vegetation	Management	Year	Mean annual temp (°C)	Rain (mm yr-1)	GPP	TER	NEE	Supplement & manure inputs	Supplement outputs	Methane from animals	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	2,600
		France	Semi-natural grassland	Grazing	1	8*	1,313*			-500			91	-410
					2	8*	1,313*			-1,120			90	-1,030
		France	Semi-natural grassland	Grazing	1	8*	1,313*			-910			54	-860
					2	8*	1,313*			-490			46	-440
		The 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	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	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
Xu and Baldocchi (2004)	Rocky silt loam - Lithic xerothents	California	Annual grassland	Grazed	2001	16.3*	567	8,670	7,350	-1,320				
					2002	16.3*	494	7,290	7,580	290				

*** averages, not specific to year of study, n.d. = no data.**

2.7 Impact of periodic cultivation on soil CO_2 flux

2.7.1 Periodic cultivation

There are many different land management practices applied to intensive pasture, all of which have the ability to increase or decrease soil C stocks. Periodic cultivation is one such practice that is increasing in New Zealand and globally (Clark *et al.* 2007; Conant *et al.* 2007). Periodic cultivation on New Zealand dairy farms has increased from once every 15 years to once every 5 – 10 years (Pasture Renewal Charitable Trust. 2009), this increase is primarily driven by a desire to increase production (Clark *et al.* 2007). Increasing cultivation frequency has the potential to reduce soil C stocks because soil aggregates are destroyed and conditions for microbial decomposition of soil C are improved (Schlesinger & Andrews 2000; Lal 2004).

Continuous cultivation has been shown to reduce soil C (Guo & Gifford 2002), however, little is known about the periodic cultivation of soil (Conant *et al.* 2007). Two recent reviews have investigated the impact of cultivation following a period of zero tillage on soil C content. Conant *et al.* (2007) used models to estimate the C loss that would occur following cultivation of a soil that had remained uncultivated for a significant period of time. The study analysed nine separate studies that investigated both immediate-term and long-term response of periodic cultivation on soil C stocks, however, none of these studies included periodic cultivation of intensively grazed grassland as all paddocks were initially under crop prior to cultivation (Conant *et al.* 2007). Govaerts *et al.* (2009) reviewed the impact that conservation agriculture has on soil C stocks and how much C is lost following rotational cultivation of no till plots. The study concluded that cultivation of no till cropland resulted in a decline in soil C, however, the reviewed studies had only sampled to a shallow depth (less than 30 cm) and previous studies have shown that this can produce unreliable results (Baker *et al.* 2007; Blanco-Canqui & Lal 2008). Both reviews concluded that the impact of periodic cultivation on soil C stocks is not well understood (Conant *et al.* 2007; Govaerts *et al.* 2009)

Studies that measured CO_2-C fluxes following a one-off cultivation event are summarised in Table 2.2. These studies differ greatly both in cultivation practices and CO_2 measurement techniques. Cultivation practices include mouldboard plough to various depths, blade rotovator and disk harrow, as well as combinations of these cultivation practices such as mouldboard plough followed by disk harrow. CO_2 fluxes were all measured using chambers, however, there can be large variation between the results of an alkali trap and a dynamic or closed chamber (see section 2.4.2) (Rochette *et al.* 1992). The study period and sampling frequency is also inconsistent across available studies.

Yamulki & Jarvis (2002) measured the impact of a one-off cultivation event on CO_2 fluxes from a clay loam over 21 days in Devon, U.K., six measurements of CO_2 flux were made and from these measurements the average daily flux from cultivated pasture was 43 kg C ha^{-1} d^{-1}. Grandy & Robertson (2006) measured the CO_2-C flux over the first 60 days following cultivation of a previously uncultivated soil, again few measurements (13) were made with the average daily flux being 54 kg C ha^{-1} d^{-1}.

In New Zealand, Aslam *et al.*(2000) measured the long term impact of land use change (pasture to crop) on soil CO_2 emission. Short-term loss for the four days following mouldboard plough (200 mm) was between 35 and 73 kg CO_2-C ha^{-1} and losses were between 26 and 81 kg CO_2-C ha^{-1} for the six days following power harrow. Again, this study involved few measurements, with 12 days of measurements made for a 341 day study period. Mudge (2009) measured CO_2 fluxes from cultivated and pasture dairy paddocks during a drought in the Waikato region of New Zealand. The study had a high measurement frequency with 20 days of measurement made over the 39 day study period and an average daily CO_2-C flux of 38 kg C $ha^{-1}d^{-1}$ from the cultivated paddocks. Despite very large differences in cultivation approaches and measurement methods daily losses are similar generally ranging from 31 kg C ha^{-1} d^{-1} (Quincke *et al.* 2007) to 68 kg C ha^{-1} d^{-1} with one exception at 131 kg C ha^{-1} d^{-1} (Reicosky & Lindstrom 1993), when this exception was excluded the mean daily CO_2-C flux from the reviewed studies was 45.9 kg C ha^{-1} d^{-1} and the median was 43 kg C ha^{-1} d^{-1}.

The paucity of data on CO_2 fluxes following periodic cultivation of long term pasture (Table 2.2) highlights the need for more studies to be performed for a wide range of soils. In New Zealand, the study by Mudge (2009) investigated the impact of a typical pastoral renewal event, however, loss was only measured from one soil type during a 100 year drought. Consequently, the application of this data to other NZ sites is limited and likely represents an extreme value which is possibly quite low as respiration was constrained by a very low soil moisture content (θ_v< 40%). Measuring the CO_2 loss from a range of other soil types following periodic cultivation events during climatically contrasting years would enable conclusions to be drawn on the impact of periodic cultivation on New Zealand's soil C stock.

Table 2.2 CO_2-C loss following one off cultivation from previous studies (adapted from Mudge (2009))

Reference	Soil	Location	Method	Treatment	Number of measurements	Study duration (days)	Daily CO_2-C flux (kg C $ha^{-1} day^{-1}$)	Total CO_2-C loss (kg C ha^{-1})
Aslam *et al*. 2000	Typic Andoaqualf	New Zealand	Alkali trap	Permanent pasture	12	341	92	31270
				Mouldboard plough to 200 mm then sown in maize in spring and oats in winter	12	341	60	20460
				No till sown in maize in spring and oats in winter	12	341	63	21313
Eriksen & Jensen 2001	Typic Hapludalt	Tjele (Denmark)	Dynamic chamber	Mouldboard plough to 220 mm	23	79	33	2600
				Untilled	23	79	18	1400
Grandy & Robertson 2006	Mesic Typic Hapludalf	Michigan (U.S.A)	Static chamber	Undisturbed field	13	60	41	2448
					10	138	30	4074
					19	198	34	6653
				Tilled once with a mouldboard plough then disc harrowed and left fallow (no spray)	13	60	54	3269
					10	138	48	6690
					19	198	43	8554
Mudge 2009	Typic Orthic Gley soil	New Zealand	Closed chamber	Spray followed by Mouldboard plough to 200 mm then power harrowed and rolled	~20	39	38	1496
				Permanent pasture	~20	39	37	1446

Table 2.2 continued

Reference	Soil	Location	Method	Treatment	Number of measurements	Study duration (days)	Daily CO_2-C flux (kg C ha^{-1} day^{-1})	Total CO_2-C loss (kg C ha^{-1})
Quincke *et al.* 2007	Mesic Typic Argiudolls	Nebraska (U.S.A.)	Closed chamber	No till (corn/soybean rotation)	~8	30	38	1149
				Chisel plow to 200 mm	~8	30	60	1809
				Mouldboard plough to 200 mm	~8	30	31	930
Reicosky & Lindstrom 1993	Aeric Calciaquoll	Minnesota (U.S.A.)	Closed chamber	Mouldboard plough to 250 mm	18	19	131	2490
				Moldboard plus disk harrow twice	18	19	68	1295
				Disk harrow to 75 mm	18	19	56	1066
				Chisel plough to 150 mm	18	19	53	998
				No till	18	19	26	499
Yamulki & Jarvis 2002	Clay loam	Devon (U.K.)	Closed chamber	Permanent pasture cut one week prior to experiment	6	21	76	1600
				Cut pasture sprayed and cultivated to 200 mm with blade rotovator	6	21	43	901

2.7.2 Variability between soil types

Under a single land use, soil C content, and C turnover time differ because of differences in soil chemical, biological and physical properties. For example, soils formed in volcanic deposits generally have the greatest amount of organic C of all the mineral soil orders (Dahlgren *et al.* 2004). Agricultural systems in New Zealand are established on a mixture of soils and the impact of physical disturbance on soil C stocks for a range of soil types is poorly understood.

In New Zealand, Parfitt *et al.* (1997) found that after 20 years of continuous cropping, total C in an allophanic soil decreased by 10 t C ha^{-1}; whereas, 23 t C ha^{-1} was lost from an adjacent non-allophanic recent soil. This led Parfitt *et al.* (1997) to conclude that allophanic clays protected soil C and reduced soil C turnover rates. Parfitt *et al.*(2002) compared C stabilisation from adjacent allophanic and non-allophanic soils under continuous maize cropping using natural variations in stable isotopes (^{13}C). After 25 years of continuous cropping, 78% of the old pasture C was retained in the allophanic soil while 69% was retained in the non-allophanic soil leading to the conclusion that allophanic soils are able to stabilise a greater proportion of “old” soil C than non-allophanic soils.

Allophanic soils stabilise soil organic C due to high concentrations of allophanic clays and Al_{py} (aluminium extractable in pyrophosphate) (Percival *et al.* 2000; Matus *et al.* 2008; Parfitt 2009). Percival *et al.* (2000) applied multiple regression analysis to soil organic C and various soil and site properties to demonstrate that Al_{py} and allophane content explained the greatest amount of variation in soil C content (Percival *et al.* 2000).

It has been shown that contrasting soil types can lose significantly different amounts of soil C following long term continuous cropping (Parfitt *et al.* 1997; Parfitt *et al.* 2002), however, there are no studies that have measured short term soil CO_2-C flux following a one-off cultivation event from adjacent contrasting soil types.

2.8 Controls on CO_2-C flux

While soil C stocks are clearly dependent on site specific factors such as land use and soil type, temperature and moisture content are considered the major controllers of soil respiration (Davidson & Janssens 2006). Recent field and lab research has shown that respiration from New Zealand's grazed pasture is predominately controlled by soil temperature when soil moisture is not limited (Brown *et al.* 2009).

2.8.1 Moisture content

The impact of soil moisture content on soil CO_2-C flux is not well understood. Numerous studies have been aimed at describing the relationship between soil water content and respiration, however, the main conclusion from these studies has been that low water contents can inhibit CO_2 production in soils (Davidson *et al.* 2000). The large majority of studies that have investigated the impact of moisture content on respiration have been conducted in forest ecosystems (Davidson *et al.* 2000) which offer little comparability to dairy pasture.

Immediately after soil is cultivated the surface can become aerated and dry. Rewetting this dry soil causes a rapid increase in soil respiration, this effect has been termed the "Birch Effect" after the work of H. F. Birch who described the impact of rewetting dry agricultural and forestry soils in East Africa during the 1950's and 1960's (Jarvis *et al.* 2007). The effect has also been observed on dairy farms; Mudge (2009) measured large pulses of CO_2 following the wetting of dry pasture during drought. The driver behind this increase in CO_2 flux is due to an accumulation of biodegradable C during the dry period, which is rapidly consumed by microbial activity following rewetting (Orchard & Cook 1983).

2.8.2 Temperature

Due to increased concern about climate change there is considerable interest in determining the control that temperature exerts on soil CO_2 flux. Numerous reviews have concluded that soil temperature has a positive relationship with soil respiration (Raich & Schlesinger 1992; Fang & Moncrieff 2001; Davidson & Janssens 2006). Senthilkumar *et al.* (2009) sampled virgin grassland soil to 20 cm in 1986 and then re-sampled the same site in 2007, the only management practice that had been applied to this ecosystem was annual mowing which started in 1960. The study found that soil C declined by 0.8 kg C ha^{-1} yr^{-1} soil over the 21 year period between 1986 and 2007 and concluded that this loss was likely due to the increase in local air temperature of 0.6 °C over the last century. It is difficult to have confidence in the conclusion of Senthilkumar *et al.* (2009) as sampling depth was shallow (0-20 cm) and sampling frequency would need to increase to identify a constant decline in soil C content (e.g. Schipper *et al.* (in press)).

It is well established that increasing soil temperature increases soil respiration, however, the form of the relationship between soil respiration and temperature is disputed, for a full review of the temperature sensitivity of soil C decomposition the reader is referred to the review by Davidson & Janssens (2006). However, by far the most common method to model soil respiration at a given temperature is the Lloyd and Taylor (1994) regression model, this model is commonly applied to partition GPP and TER fluxes from measured NEE at EC sites (Reichstein *et al.* 2005). The Lloyd and Taylor (1994) regression model is based on an Arrhenius type equation where effective activation for respiration varies inversely with temperature. The relationship has been widely applied and has recently been used in a New Zealand study of factors regulating soil surface respiration in grazed pasture (Brown *et al.* 2009).

Brown *et al.* (2009) studied the regulation of soil surface respiration from a grazed pasture in New Zealand and use the Lloyd & Taylor (1994) equation to model the relationship between soil respiration and temperature while a linear model was used to describe the relationship between respiration and soil moisture. Brown *et al.* (2009) showed that soil respiration was predominantly controlled by soil temperature because soil moisture was rarely low enough to limit soil respiration during the study period.

2.9 Water use efficiency

Ecosystem water use efficiency (WUE) is the ratio of the daily integrated GPP and the daily integrated evaporation (Tong *et al*. 2009), the ratio is important for indicating the relationship between C gain and H_2O loss. Ecosystem WUE differs from plant WUE in that all components of the ecosystem are included (e.g. evaporation from bare soil and interception water loss) whereas these sources would not be included in the calculation of plant WUE. The determination of ecosystem WUE can enhance our ability to predict how climate cycles and climate change may regulate the C and energy budgets of different ecosystems. WUE also provides understanding of the tolerance of different pasture species to external factors such as climatic stress or changing land management. WUE at the individual leaf scale is well understood, however, the measurement of whole ecosystem WUE is less understood with the large majority of studies focussing on WUE of forest and cropland (Tong *et al.* 2009) rather than grazed pastures. Table 2.3 lists the studies which have investigated the WUE of pasture and crops

Table 2.3 Summary of water use efficiency of pasture, natural grassland and crops. All studies excluding Schapendonk *et al.* (1997) determined WUE using eddy covariance techniques. WUE refers to the annual average ecosystem WUE unless stated otherwise (Hunt *et al.* 2002; Law *et al.* 2002).

Reference	Location	Vegetation	Management	Year	Mean annual temp (°C)	Rain (mm yr^{-1})	WUE (g C kg^{-1} H_2O)
Aires *et al.* (2008)	Portugal	Annual C3 and perennial C4 grasses	Grazed by sheep	2005	14.7	364	3.3
				2006	14.5	751	4.3
Hunt *et al.* (2002)[a]	New Zealand	Tussock	Grazed by sheep	2002	12.0	646	0.3
Law *et al.*(2002)[b]		Natural grassland	Natural grassland	2002			0.9
Ponton *et al.* (2006)	Alberta, Canada	Natural grassland	Natural grassland	2005	5.7	401	1.8
Schapendonk *et al.*(1997)[c]	The Netherlands	Perennial rye grass	Harvested	1993-1995	14.2	Irrigated	3.5

[a] Study period was 212 days
[b] Summary of growing season WUE from various FLUXNET sites, refer to www.eosdisornl.gov/FLUXNET for further information
[c] Study was conducted in laboratory environment using soil monoliths

The majority of studies from Table 2.3 focused on non-managed natural grasslands, such as the study by Law *et al.* (2002) who summarised the mean monthly WUE during the growing season of FLUXNET sites set up over natural grassland ecosystems and found that WUE ranged from 0.027 to 1.6 g C kg^{-1} H_2O. Grazed pasture is poorly represented in Table 2.3 and WUE from intensively managed dairy pasture has not been measured. Aires *et al.* (2008) measured ecosystem WUE of 3.3 g C kg^{-1} H_2O during a dry year and 4.3 g C kg^{-1} H_2O during an average year from mixed pasture grazed by sheep in Portugal. These values of WUE exceed the range presented by Law *et al.* (2002) but agree with the WUE of 3.5 g C kg^{-1} H_2O presented by Schapendonk *et al.* (1997) for ryegrass pasture grown in a controlled laboratory environment. Measurements of ecosystem scale WUE are needed for intensive pasture to improve current understanding of CO_2 and H_2O exchange in these managed ecosystems.

2.10 Summary

This review has highlighted the need for increased research into the C dynamics of intensively grazed pasture ecosystems. Most NECB measurements of pasture made internationally do not apply to New Zealand pasture because of largely different management practices. Long term NECB measurements need to be made for intensively grazed pasture under contrasting climatic conditions which account for all inputs and exports. Current research has measured NECB for extreme conditions such as organic soils (Nieveen *et al.* 2005) and dry conditions (Mudge 2009), but few studies into high intensity grazing on mineral soils under average climatic conditions exist in New Zealand or indeed globally.

The impact of periodic cultivation on soil C has yet to be well defined, with reviews calling for an increase into the effects of single cultivation events. Even fewer studies have examined the cultivation of pasture systems. Previous studies have shown that soil C loss is significantly different between contrasting soil types under continuous cultivation but whether differences also occur following one-off cultivation events is not known.

Currently, there is adequate understanding regarding WUE at the leaf scale, however, the use of eddy covariance provides the opportunity to measure WUE at the ecosystem scale. These large scale measurements have previously been made over forested and cropped land but few studies have investigated grassland, and even fewer have studied intensively grazed pasture.

CHAPTER THREE

Annual carbon balance of dairy pasture

3.1 Introduction

In order to understand changes in soil carbon (C) content, annual C balances need to be constructed for a range of ecosystems during a variety of climatic conditions. One ecosystem that has received little attention but occupies a significant area of the Earth's terrestrial area is grazed pasture. Grazed pasture occupies about 39% of New Zealand's total land area (Ministry for the Environment 2007) and globally, grazed pasture accounts for about 26% of the ice free land surface (Steinfeld *et al.* 2006).

Land under pasture fixes CO_2 through photosynthesis and loses CO_2 through respiration derived from autotrophic and heterotrophic sources. Carbon can be imported to grazed pasture in the form of feeds or supplements which have been produced outside the farm boundary and C can be exported from a farm as product (vegetation biomass, meat, milk or wool), methane emission, leaching and erosion. These C inputs and outputs must be accounted for to create a complete C balance.

Understanding the C balance of grazed pasture systems is of high importance as previous research has identified that the stores of soil C under pasture may be declining (Schipper *et al.* 2007; Meersmans *et al.* 2009). A review of the literature (Chapter 2) identified that there is a lack of information on the C balance of intensively grazed soils. To improve understanding of C cycling in pasture, measurements of C fluxes need to be made.

Eddy covariance (EC) can be used to continuously measure large scale net ecosystem CO_2 exchange (NEE). Adding other C inputs and outputs to NEE allows the net ecosystem C balance (NECB) to be created (Chapter 2, section 2.6.2). In New Zealand, a NECB was constructed for an intensively grazed pasture using EC for a dairy pasture on a peat soil (Nieveen *et al.* 2005), however, peat soils will likely behave very differently to mineral soils. Once drained for productive use peat soils lose large amounts of CO_2 through mineralization. A NECB was also established for a dairy pasture on a mineral soil during a year which experienced a one in 100 year drought (Mudge 2009) and this ecosystem had a C gain of 199 ± 500 kg C ha^{-1} yr^{-1}. Further studies of NECB need to be conducted so that inter-annual variation in the C balance of these ecosystems can be identified.

EC can also be applied to measure ecosystem water use efficiency (WUE) which is the ratio of C fixed via photosynthesis to moisture leaving the soil via evaporation (*E*) (Aires *et al.* 2008). The determination of ecosystem WUE can enhance our ability to predict how climate change might alter the C and energy budgets of grazed pastures. WUE at the individual leaf scale is well understood, however, ecosystem scale studies have primarily focused on determining WUE of forest and cropland (Tong *et al.* 2009) with few studies investigating the WUE of intensively grazed pasture. Research into ecosystem scale WUE of a typical New Zealand dairy pasture would provide increased knowledge into the relationship between H_2O and C cycling within this ecosystem.

The objective of this chapter is to create a net ecosystem C balance (NECB) for an intensively grazed pasture site for 2009 and analyse inter-annual variability in NEE, NECB and WUE between 2008 (15/12/07 – 14/12/08) and 2009 (15/12/08 – 15/12/09). Eddy covariance data were collected by Mudge (2009) for 2008. These data are also used here in combination with data for 2009, however, the data have been reprocessed using a different combination of raw data filtering and gap filling methods. Mudge (2009) did not determine WUE.

3.2 Methods

3.2.1 Site description

A 3 m high 40 cm triangular lattice tower was erected to support an open path EC system in open pasture at DairyNZ’s Scott Farm (NZMS 260 S14 271E 637N), Waikato, New Zealand during December 2007. The tower was located approximately 200 m from the eastern and western farm boundaries, 250 m from the southern boundary and more than 500 m from the northern boundary, land use outside of this farm boundary was also largely grazed pasture (Figure 3.1). The paddocks surrounding the EC tower were primarily Matangi silt loam soil which is classified as a Typic Orthic Gley soil (Hewitt 1998). Plant cover was predominantly perennial ryegrass (*Lolium perene*) and white clover (*Trifolium repens*). The paddocks surrounding the EC tower were rotationally grazed by small herds (3 cows ha^{-1}) throughout the entire study period. For photographs of instrumentation and the site, refer to digital Appendix E.

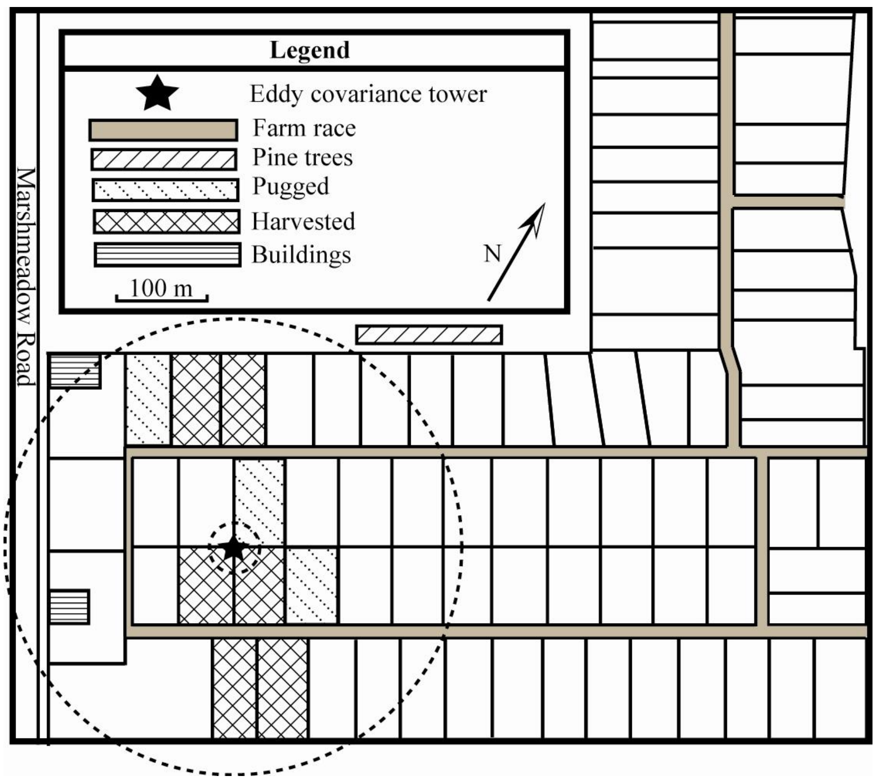

Figure 3.1 Map of Scott Farm showing the location the EC tower and surrounding paddocks that were harvested (19/10/2009) and pugged by cows (13/7/2009). The outer circle centred on the EC tower represents the typical area from which 80% of day time fluxes were derived, while the inner circle represents the typical peak of the daytime flux foot print. Adapted from Scott Farm map created by DairyNZ.

3.2.2 Soil moisture release characterisation

A soil pit was excavated approximately 15 m north west of the EC tower in August 2009, soil cores were extracted from each soil horizon to a depth of one metre (McQueen 1993). Moisture release was determined from the cores following the method outlined in McQueen (1993) at Landcare Research, Hamilton to obtain full moisture release characterisation for the Matangi silt loam (see section 3.3.2 for results and digital Appendix E for raw data).

3.2.3 Instrumentation

The EC measurement system comprised of a 3D sonic anemometer (CSAT3, Campbell Scientific Inc, Logan, UT, USA) and an open path infrared gas analyser (IRGA) (LI-7500, LI-COR Inc., Lincoln, NE,USA), these instruments were installed at 2.84 m height on the EC tower. The IRGA was installed horizontally to reduce the time that water remained on the IRGA windows following precipitation, this mounting position allowed a greater volume of quality data to be collected. The IRGA was calibrated about once every four months throughout the two year study period.

Air temperature and humidity were measured (HMP45A, Vaisala, Finland) at 3 m, on 17/2/09 an aspirated shield (43502, R.M. Young, Michigan, USA) was installed to reduce possible radiative heating and cooling. Global (shortwave) radiation (Rg) (LI-190SB, LI-COR Inc), photosynthetic photon flux density (PPFD) (LI-190SA, LI-COR, Inc.) and wind direction (W200P, Vector Instruments, Clwyd, UK) were measured at 3 m height. Wind speed (A101M, Vector Instruments, Clwyd, UK) and net radiation (Q6.7.1, Radiation and Energy Balance Systems, Seattle, USA) were both measured at 0.93 m height, net radiation was corrected for windspeed following manufacturer's instructions. Soil heat flux was measured using heat flux plates (HFP01SC, Hukseflux, Delft, The Netherlands) at a depth of 60 mm. A tipping bucket rain gauge was used to measure precipitation (TB5, Hydrological Services, Florida, USA) at 0.4 m above the soil surface. Soil temperature was measured at 50 mm and 100 mm using soil thermisters (100K6A1B Farnell, Auckland, New Zealand). Volumetric soil moisture content was measured at 50 mm and 100 mm using water content reflectometers (CS616, Campbell Scientific Inc., Logan, UT, USA). Instruments were powered by four north facing solar panels (SX-80U, BP Solar) which charged a bank of four 12 V batteries (R220, Hella-Endurant, Lippstadt, Germany) connected to the data loggers and instruments.

3.2.4 Data loggers

Campbell Scientific data loggers received the electronic output from each instrument. A CR3000 data logger (Campbell Scientific Inc, Logan, UT, USA) acquired measurements from the 3D sonic anemometer, IRGA and some supporting instruments at a frequency of 20 Hz, these high frequency data were stored on a CF memory card which was replaced every two weeks. Net radiation, PPFD, humidity, precipitation and air and soil temperature were also measured at a frequency of 20 Hz and stored in the CR3000 as 30 minute averages or totals in the case of precipitation and sent via an automated telemetry system to the University of Waikato on a daily basis. Supporting measurements were logged to a CR10X data logger (Campbell Scientific Inc, Logan, UT, USA), which converted high frequency measurements to half hourly averages and downloaded manually to the University of Waikato computer system about once every two weeks.

3.2.5 Processing data

Data collected by the CR3000 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). Raw data were corrected weekly using a custom software program (Micrometlab) which ran on a Matlab 7.6 platform (The Mathworks Inc., Natick, MA, USA) so that it could be used for analysis.

3.2.6 Filtering and gap filling

Data quality decreased during periods of rain, frost, fog and low wind speed. Rain, frost and fog resulted in water or ice forming on the IRGA windows while rain drops falling through the IRGA optical path produced high-frequency spikes that reduced flux data quality. Low wind speeds resulted in insufficient turbulent mixing (Baldocchi 2003).

To filter out poor quality data the high frequency (20 Hz) automatic gain control (AGC) signal was used. A running AGC baseline was established and deviations from this baseline of less than negative one and greater than positive four were excluded. Data were also rejected during times of low wind-speed as the EC technique is un-reliable in these conditions, a threshold of 0.1 m s^{-1} was applied for frictional velocity (u_*), when u_* fell below this value, CO_2 exchange data was excluded. The dataset also required hard spike filtering between July and December 2009, during this period CO_2 fluxes greater than 13 µmol m^{-2} s^{-1} were excluded from the dataset submitted for gap filling. Following filtering, 41% of the data remained, much of the discarded data was due to a winter-spring period that experienced numerous rain fall events and low wind speeds (20/07/09 – 1/12/09). Although the amount of good data was low compared to the expected average data quality of 65% reported by Falge *et al.* (2001), Nieveen *et al.* (2005) recorded 45% good data at a nearby dairy pasture site which suggests the low data quality may be due to the lowland topography of the Waikato region which receives little wind and numerous rain events.

Gaps in meteorological and EC data were filled using the online gap filling model of Reichstein *et al.* (2005). The online model is based on an algorithm that first estimates the temperature sensitivity of ecosystem respiration from short-term periods, and then applies this short-term temperature sensitivity to extrapolate the ecosystem respiration from night time to day time. This algorithm allows variations in seasonal temperature sensitivity to be represented (Reichstein *et al.* 2005). NEE data were gap filled using this model then partitioned into GPP and TER using the online partitioning model of Reichstein *et al.* (2005), briefly, night-time NEE was used to predict daytime TER using the Lloyd and Taylor (1994) regression model and GPP were then calculated by subtracting NEE from TER. See Appendix C for a full explanation of the online gap filling model and flux partitioning model. Latent heat flux density (λE) data for use in the calculation of WUE (GPP÷E) from Scott Farm were gap filled using the online model of Reichstein (2005). Non-gap filled half hourly values of λE and sensible heat flux (H) used to calculate the energy balance closure for both years following the methods established by Kuske (2009), refer to Appendix D for plots of energy balance closure for 2008 and 2009.

3.2.7 Footprint analysis

The footprint of the EC tower was determined using the method of Schuepp *et al.* (1990) (Figure 3.1). Analysis was conducted using 30 minute average sensible heat flux, three dimensional sonic anemometer and meteorological data. During daylight, the distance to the peak of the flux foot print was on average 27 m compared to 52 m at night. On average, 80% of flux measurements at the study site were sourced from an area with a radius of 245 m during the day and 470 m at night. Fetch during the day was appropriate for the site, however, the footprint radius of 470 m at night exceeded the farm boundary. Land use outside of the farm boundary was largely pastoral agriculture so it is assumed that fluxes sourced from this area were representative of the farm.

3.3 Results

3.3.1 Climate

Climate and soil variables measured at the EC site (15/12/07 – 15/12/09) are presented in Figure 3.2. Total annual rainfall, air temperature and global radiation at Scott Farm can be compared to long term measurements at Ruakura climate station, approximately 6 km from Scott Farm.

3.3.1.1 Climatic conditions 2008

Total rainfall for 2008 was 1263 mm measured at Scott Farm between 15 December 2007 and 14 December 2008 which was greater than the 30 year (1979 – 2008) average of 1127 mm measured at Ruakura (NIWA 2009), however, the distribution of rainfall during the year was unusual (Figure 3.2). January was very dry with the least amount of rain being recorded for that month since records began (4.2 mm), and rainfall in February and March was also much lower than the monthly mean. Winter was wetter than normal, with rainfall for June, July and August exceeding the long-term monthly average (Figure 3.2).

Mean annual temperature for 2008 was 14.2°C, which was greater than the 30 year mean (1979 – 2008) of 13.5°C. Maximum mean monthly air temperature was 20.3 °C recorded in January, which was 2 °C greater than the 30 year average for this month (NIWA 2009). Minimum mean monthly temperature was 9 °C in July, which was the same as the 30 year average for this month. Temperatures between December 2007 and April 2008 were all above average, while May was 1.6 °C below average (11.8 °C) and mean monthly air temperatures for June – December 2008 were very close to the 30 year average air temperature for these months (NIWA 2009).

Global radiation (Rg) was greatest during the summer months (December – February) with maximum monthly Rg being recorded for January (23.7 MJ m^{-2} d^{-1}) which was 4.9 MJ m^{-2} d^{-1} greater than the 10 year average (1999 – 2008) recorded at Ruakura for January. Global radiation then started to decline during autumn and winter, recording a minimum monthly Rg of 6.0 MJ m^{-2} d^{-1} for July, which was 1.0 MJ m^{-2} d^{-1} lower than the 10 year average for this month. Overall, average daily Rg for 2008 (14.3 MJ m^{-2} d^{-1}) was similar to the 10 year average daily Rg (14.2 MJ m^{-2} d^{-1}) measured at Ruakura (NIWA 2009).

Volumetric soil moisture content declined between December and February due to the combination of above average temperatures and below average rainfall. Volumetric soil moisture content on 27 December 2007 was 48%, and then declined to 21% which was below the permanent wilting point (25 %) on the 9th of February. The soil remained dry until mid April when a significant quantity of rain fell and recharged soil moisture.

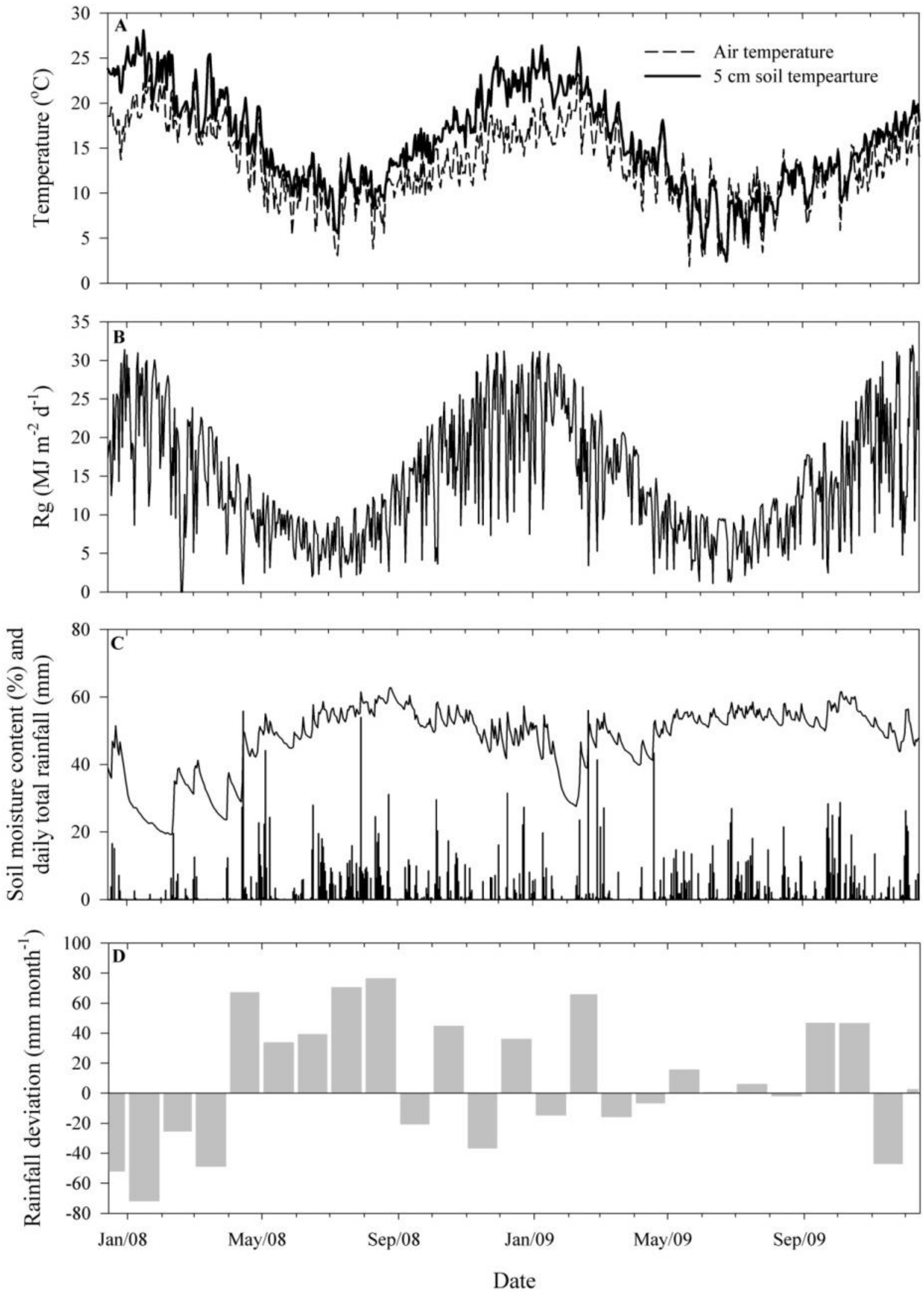

Figure 3.2 Climatic and soil variables recorded at Scott Farm during the study period (15/12/07 – 15/12/09); A) air temperature and 5 cm soil temperature. B) global radiation (Rg). C) 5 cm soil moisture (θ_v) (line) and daily rainfall (bars). D) deviation of total monthly rainfall from 30 year monthly mean (1979 – 2008) measured at Ruakura.

3.3.1.2 Climatic conditions 2009

Total annual rainfall for 2009 was 1311 mm which was similar to the 30 year mean (1979 – 2008) of 1127 mm. Rainfall was below the 30 year monthly mean in January, March, and April. Total rainfall during May, June and July was not substantially different to the long term mean for these months. During August rainfall was close to average, however, September and October both exceeded the long-term mean. Rainfall declined below the mean in November and was slightly below the 30 year mean in December (Figure 3.2).

Mean annual temperature for 2009 was 12.8°C which was lower than the 30 year mean of 13.5°C. The greatest mean monthly temperature was recorded in February (19.3°C) while minimum mean monthly temperature was recorded in July 2009 (8.9°C). February and August had monthly mean temperatures greater than the monthly 30 year average. However, in general mean monthly temperature for the rest of the year was lower than the 30 year average. Winter (June, July, August) 2009 was the coldest winter within the 30 year dataset (NIWA 2009) with mean monthly temperature in June and July being 2.5°C and 0.9°C lower than the 30 year averages respectively.

Global radiation (Rg) was greatest during the summer months (December – February) with maximum monthly Rg being recorded for January (25.0 MJ m^{-2} d^{-1}) which was 6.2 MJ m^{-2} d^{-1} greater than the 10 year average (2000 – 2009) recorded at Ruakura for the month (NIWA 2009). Global radiation then started to decline during autumn and winter, recording a minimum monthly Rg of 6.9 MJ m^{-2} d^{-1} for June, which was 0.6 MJ m^{-2} d^{-1} greater than the 10 year average for this month. Overall, average daily Rg for 2008 (14.5 MJ m^{-2} d^{-1}) was similar to the 10 year average daily Rg (14.2 MJ m^{-2} d^{-1}) measured at Ruakura (NIWA 2009).

Volumetric soil moisture content was greatest in winter due to the surplus of rainfall and low *E* rate, soil moisture declined during summer (December – February), however, soil moisture did not drop below the permanent wilting point (25 %) for the Matangi silt loam.

3.3.1.3 Climatic variation

In summary, 2008 experienced a summer (December – February) and autumn (March – May) drought followed by a wet winter. Summer during 2009 was dry, however, soil moisture deficit did not occur for longer than a few days, winter 2009 was the coldest in the 30 year record and received less rainfall than 2008. Neither year could be described as an "average" year, however, average conditions are difficult to define at this site as climate is variable from year to year.

3.3.2 Soil moisture release

Averaged moisture release characterisation for the Matangi silt loam at Scott Farm is displayed in Table 3.1. Total available water capacity for the Ap horizon was 28.2%, readily available water capacity was 10.7%, field capacity was 53.6% (5 kPa), permanent wilting point or lower limit of total available water was 25.4% (1500 kPa) and the lower limit of readily available water (100 kPa) was 42.9 %. The lower limit of total available water and lower limit of readily available water measured for the Matangi silt loam are similar the average lower limit of total available water (24%) and readily available water (43%) reported for similar Waikato soils by Singleton (1991).

Table 3.1 Moisture release characterisation and available water capacities for the Matangi silt loam at Scott Farm.

Depth (cm)	VWC (%, v/v) at applied tension (kPa)						AWC	
	5 kPa	10 kPa	20 kPa	40 kPa	100 kPa	1500 kPa	Readily (%, v/v)	Total (%, v/v)
0 - 25	57.0	53.6	49.9	47.1	42.9	25.4	10.7	28.2
25 - 32	48.7	46.1	42.0	37.9	31.7	21.0	14.4	25.1
32 -95	50.6	48.0	41.9	37.0	29.9	14.7	18.1	33.3
95 - 110+	15.9	13.5	11.4	9.8	8.1	4.5	5.4	9.0

VWC = Volumetric water content, AWC = Available water capacities.

3.3.3 Net ecosystem exchange

Running cumulative net ecosystem CO_2 exchange (NEE) for both years is presented in Figure 3.3, the site was a greater sink of CO_2 in 2009 than in 2008. Inter-annual difference in cumulative NEE was 1,068 kg C ha^{-1} yr^{-1}.

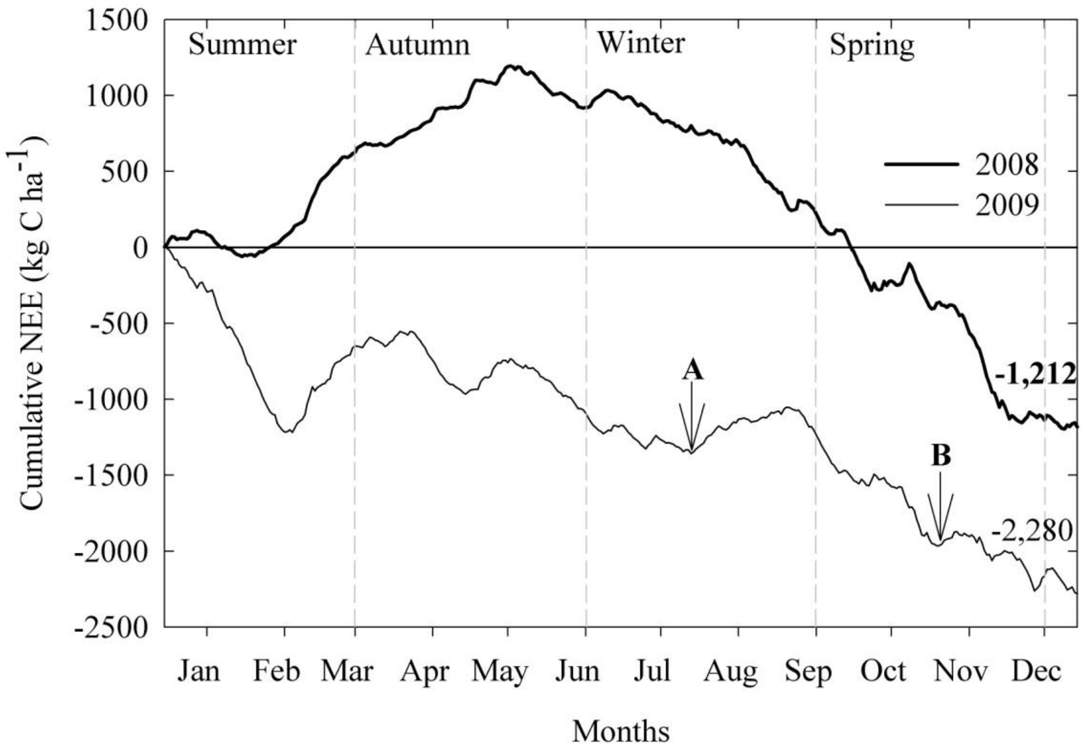

Figure 3.3 Cumulative NEE for 2008 and 2009, two management events were observed during 2009 (A & B) and are expanded on in section 3.3.3.4.

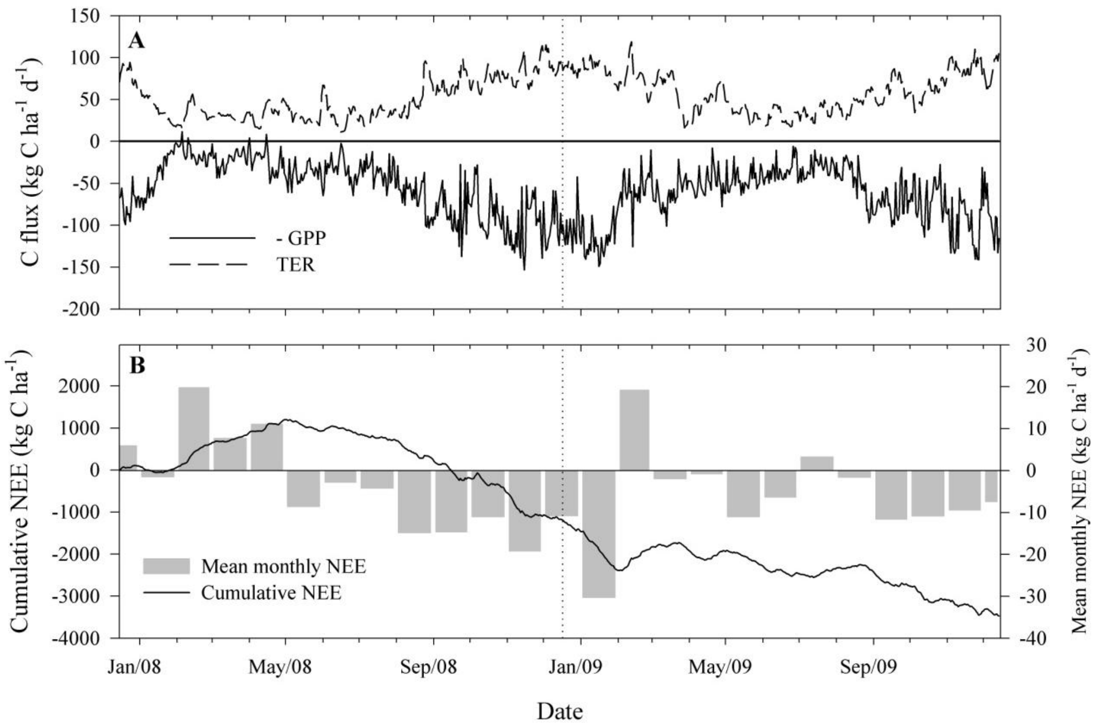

Figure 3.4 A) Daily sum of GPP and TER, GPP is represented as negative in this Figure to enable GPP to be distinguished from TER. B) Continuous cumulative NEE (line) and mean monthly NEE (bars) for the two year study period (15/12/07 – 15/12/09). The vertical dotted line in both figures represents the division between 2008 and 2009 data.

3.3.3.1 NEE 2008

Scott Farm was a net sink of CO_2 for 2008 (15/12/07 – 15/12/08) with cumulative NEE being -1,212 kg C ha^{-1} (Figure 3.3). NEE for 2008 was initially positive making the site a slight source of CO_2 during December 2007, the site then became a slight sink of CO_2 during January 2008 before drought conditions caused GPP to practically cease while TER continued at a decreased rate resulting in the site becoming a source of C (Figure 3.4). The site remained a source of CO_2 from February until May when the drought ended causing GPP to exceed TER (Figure 3.4). NEE was then negative for the remainder of the 2008 study period indicating the site was a sink of CO_2.

3.3.3.2 NEE 2009

Scott Farm was a net sink of CO_2 for 2009 (15/12/08 – 15/12/09) with cumulative NEE being -2,280 kg C ha^{-1} (Figure 3.3). Scott farm was a CO_2 sink between December 2008 and January 2009. A rapid switch in NEE occurred between January and February, the site was a strong sink of CO_2 in January while the site was a strong source of CO_2 in February (Figure 3.4). The cause of this switch is displayed in Figure 3.4, GPP declined rapidly in February while TER remained relatively static which resulted in site being a source of CO_2. The site was initially a weak CO_2 sink following February with GPP and TER following a similar pattern between May and June as soil moisture recovered and photosynthesis increased. NEE was positive in July making the ecosystem a source of CO_2 (Figure 3.4), however, from August onwards NEE was negative and the site was a CO_2 sink.

3.3.3.3 Variation in NEE

Cumulative NEE differed between 2008 and 2009 by 1,068 kg C ha^{-1}. Scott Farm was a source of C from the start of the study in December 2007 until May 2008 when the site had recovered from the effects of drought. During the same period in 2009 (December 2008 – May 2009) the site was largely a net sink of CO_2, however, a dry period during February caused NEE to become positive (Figure 3.4). Scott Farm was a greater sink of CO_2 following winter 2008 (June – August) as winter 2009 was the coldest recorded in the last 30 years which caused the spring growing season to be delayed. The 2008 spring growing season began in early August while the 2009 spring growing season was delayed until late August early September, this delay caused by the cold winter has likely reduced potential cumulative NEE.

Figure 3.5 A) Example of pugged paddock one day after cows removed (14/7/09), B) and eight days after cows removed (20/7/09).

3.3.3.4 Land management

During 2009, the site often changed on a daily basis from a CO_2 source (positive NEE) to a CO_2 sink (negative NEE). Two management events were observed within the flux foot print (Figure 3.1) which altered the above ground biomass and could have subsequently caused NEE to become positive. These events were pugging and harvesting which are displayed in Figure 3.3 as event A and event B respectively.

The pugging event occurred on 13 July, when about 20 cows where housed within 0.5 ha paddocks (Figure 3.1), many of these cows were pregnant and the paddock was heavily pugged damaging the soil and vegetation (Figure 3.5). Following the pugging event, the site became a source of CO_2 (Figure 3.6).

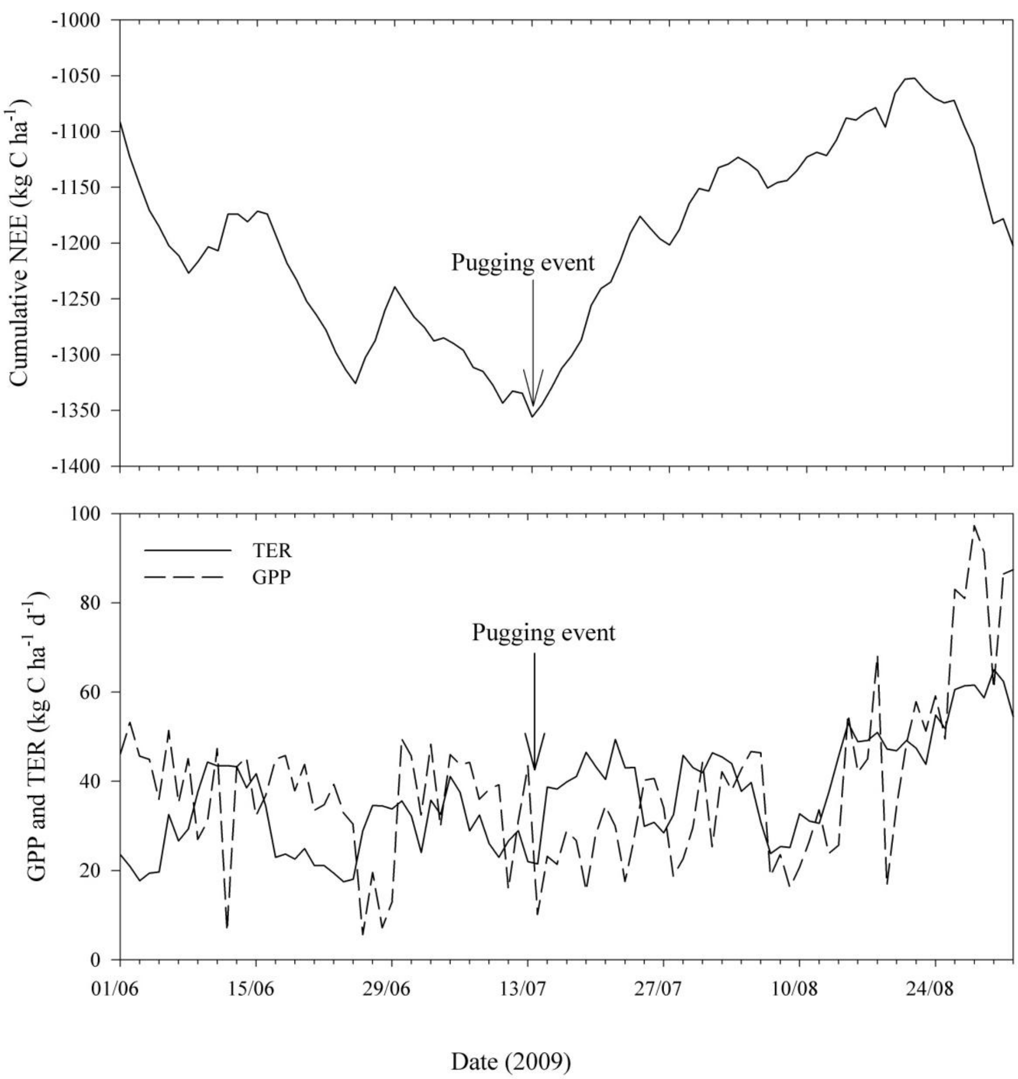

Figure 3.6 Cumulative NEE and daily GPP and TER for the pugging event.

A second event was the harvesting of dry matter which was observed on 19 October 2009 from the paddocks shown in Figure 3.1. Although this event was observed it was difficult to identify any clear response in the cumulative NEE (Figure 3.7).

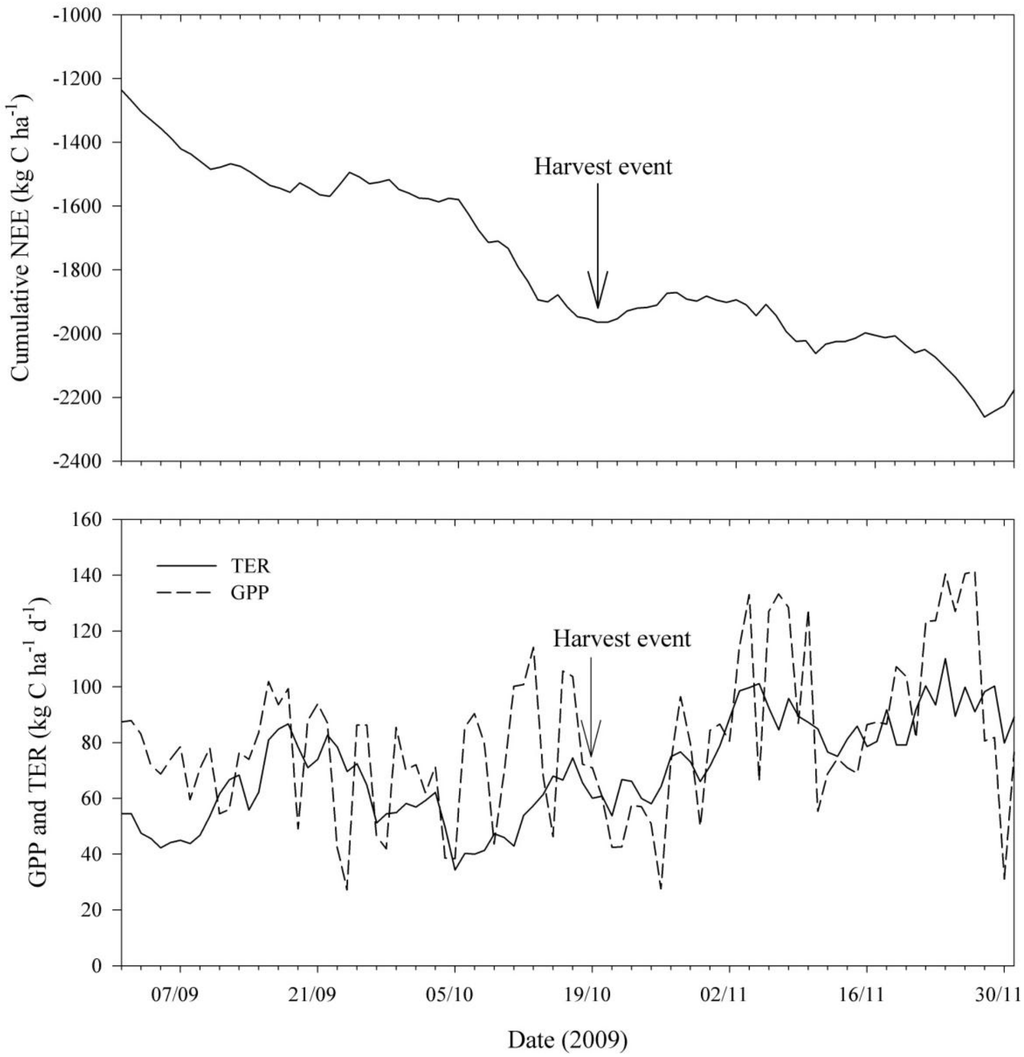

Figure 3.7 Cumulative NEE and daily GPP and TER for the harvest event.

Although these specific events were observed in the field, concluding that these management events forced measured fluctuations in NEE is difficult because in both circumstances land management change was only applied to a small area of the flux foot print (Figure 3.1). Following pugging GPP declined slightly while TER increased slightly (Figure 3.6), the variability in GPP and TER around the observed pugging event may have driven the positive NEE rather than the physical affect of pugging. Following the harvest event TER increased while GPP declined initially before recovering (Figure 3.7), again the variability of these measurements makes identifying the impact of this land management practice on NEE difficult. No sudden change in temperature or moisture content was observed (Figure 3.8).

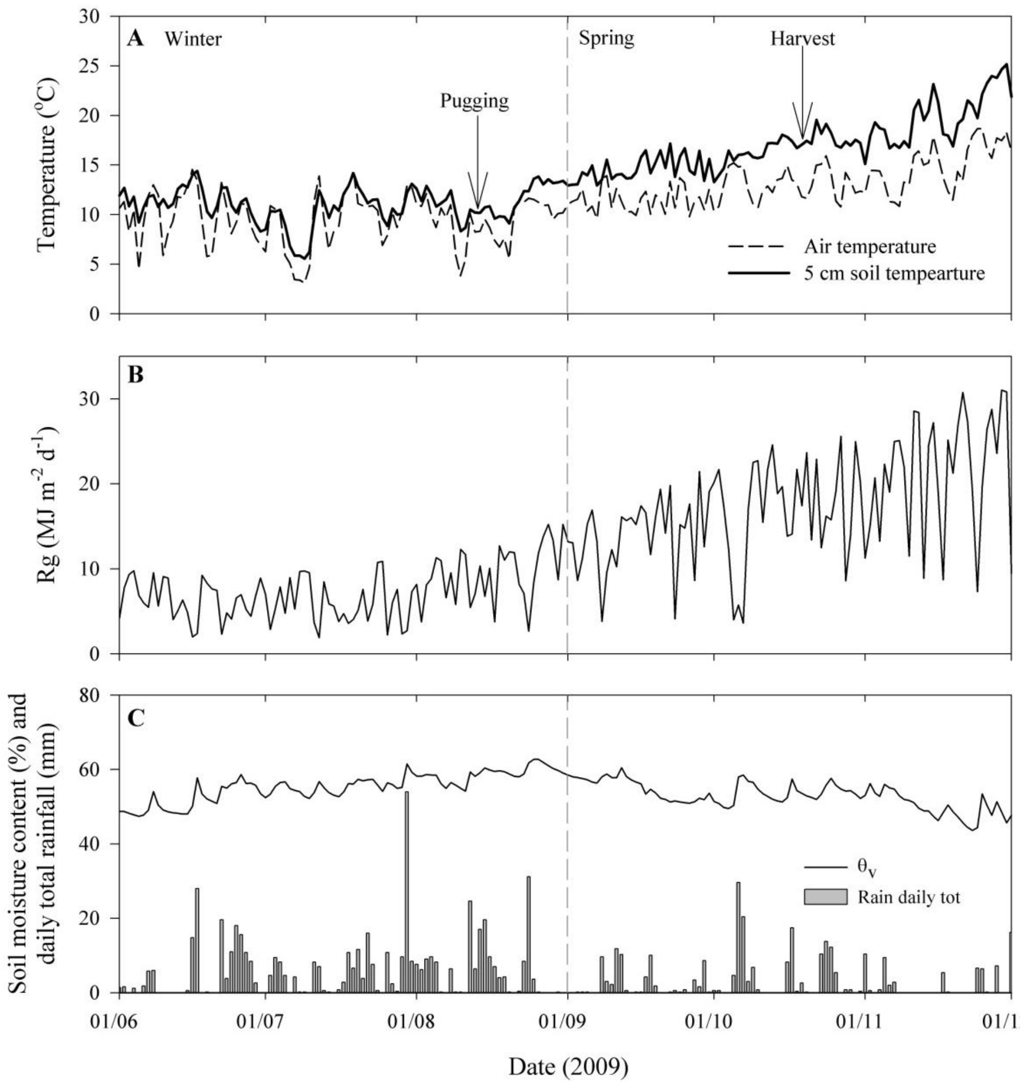

Figure 3.8 Climate variables for pugging (13/7/09) and harvest event (19/10/09). A) Air temperature and 5 cm soil temperature, B) global radiation (Rg), C) soil moisture content and daily total rainfall.

3.3.4 Annual net ecosystem carbon balance

Annual NECB for the site was compiled following the method of Chapin *et al.* (2006) and Soussana *et al.* (2007) (see Chapter 2, section 2.3.1, equation 2.1). Table 3.2 compares the final net ecosystem C balance for both years. Cumulative NEE for 2009 was -2,280 kg C ha^{-1}, GPP was 23,895 kg C ha^{-1} and annual TER was 21,615 kg C ha^{-1}.

GPP at this site is similar to the estimate made from average dry matter production for an average New Zealand dairy system (21,818 kg C ha^{-1}) in Chapter 2 (see Section 2.5.1.1). Mudge (2009) collected EC and meteorological data between 15 December 2007 and 14 December 2008, these data were reprocessed for this study. For the same site in 2008 cumulative NEE was -1,212 kg C ha^{-1}, GPP was 19,244 kg C ha^{-1} and TER was 18,032 kg C ha^{-1}. Records of feed import, pasture harvest and product exports were not available for Scott Farm from May 2009 – December 2009 and were estimated. During 2009, feed import was zero as silage was made on site and returned to the harvested area during winter. During late January 2009, grass was cut for silage from the flux footprint and exported to DairyNZ's Lye Farm which resulted in a net loss from Scott Farm of 98 kg C ha^{-1} for the year. Average MS production for 2009 was estimated at 841 kg C ha^{-1} yr^{-1} (E. Thom, pers comm., 2009[1]; C. Roach, pers comm., 2009[2]). Stocking rate was 3 cows ha^{-1} throughout the study and if methane loss per livestock unit is assumed to be 0.267 kg CH_4-C cow^{-1} day^{-1}, which is the average emission from three recent studies (Robertson & Waghorn 2002; Laubach & Kelliher 2004; Woodward *et al.* 2004), then total CH_4-C loss from livestock during the study period was 292 kg C ha^{-1} yr^{-1}. Carbon leached from the soil was not measured but has been estimated to be approximately 50 kg C ha^{-1} from the values produced by Sparling *et al.* (2006) who measured leaching of dissolved organic C of 56 kg C ha^{-1} yr^{-1} from a gley soil similar to the Matangi silt loam. Erosion of C from this site was assumed to be negligible as the site was flat and did not become un-vegetated during the study. Manure input directly from cows and subsequent decomposition is assumed to be part of the internal cycling component of measured NEE, along with soil, grass and cow respiration.

When all imports and exports of C are accounted for (Table 3.2) the NECB of Scott Farm between 15 December 2008 and 15 December 2009 was -999 kg C ha^{-1} yr^{-1}. NECB for the same period in 2008 (15/12/07-14/12/08) was -199 kg C ha^{-1} yr^{-1}. Overall, the site was a greater sink of C in 2009 than 2008.

[1] Personal communication with Dr. Errol Thom, Scientist, DairyNZ, Hamilton.
[2] Personal communication with Chris Roach, DairyNZ, Hamilton.

Table 3.2 Comparison of net and gross C fluxes from Scott Farm for 2008 recalculated from (Mudge 2009) and 2009, units are kg C ha^{-1} yr^{-1}.

Flux	2008	2009
NEE	-1,212	-2,280
GPP	19,244	23,895
TER	18,032	21,615
Feed import	218	0
Milk solids	840	841
Silage export	70	98
Methane	271	292
Erosion	0	0
Leaching	~50	~50
NECB	**-199 ± 500**	**-999 ± 500**

The difference between 2008 and 2009 in NECB was 800 kg C ha^{-1}. An estimated error bound of ± 500 kg C ha^{-1} was applied to the NECB values (see following section).

3.3.4.1 Errors associated with EC measurements

There are two common sources of error associated with the measurement of NEE using EC, these are; night time measurements and energy balance closure (Baldocchi 2003). There is also a large random error component associated with short term measurement of NEE (typically half hour scale). The uncertainty at these short time scales can be large (Hollinger *et al.* 2004), however, it has been determined that this uncertainty decreases dramatically with time and the contribution of this error is less than 10% at an annual scale (Hagen *et al.* 2006).

At night, fluxes can be underestimated due to a combination of insufficient turbulent mixing, incorrect measurement of CO_2 in the air space and soil, and the drainage of CO_2 out of the canopy volume at night (Baldocchi 2003). To account for this underestimate, filtering and gap filling were performed as described in Section 3.2.7 and Appendix C.

Studies frequently find that the sum of latent and sensible heat exchange, measured with the EC technique, do not match with independent measurement of available energy (Baldocchi 2003). Tests of surface energy balance closure suggest that turbulent fluxes at most EC sites are 10 – 30% too small to close the energy balance which raises the possibility that CO_2 fluxes are also underestimated (Wilson *et al.* 2002; Baldocchi 2003). The discrepancy can be corrected by adjusting CO_2 flux densities in proportion to the underestimated energy balance closure, however, this approach involves reliance on measurements of global radiation and soil heat flux from instruments with small footprints mounted directly beneath the tower, whereas EC measurements at Scott Farm represent an area of between 20 and 70 ha. The energy balance ratio at Scott Farm was calculated to be underestimated by 12% in 2008 and by 9% in 2009 which is comparable to previous EC studies (Wilson *et al.* 2002) (refer to Appendix D for plots of half hourly energy balance closure). This may have caused fluxes of CO_2 to be underestimated by a similar magnitude. The effect that this underestimation has had on the raw components of ecosystem WUE (GPP and λE) is assumed to be negligible as both components are assumed to be underestimated by a similar magnitude. The current study did not correct for energy balance closure, which is comparable to the approach applied by recent EC studies (Nieveen *et al.* 2005; Ammann *et al.* 2007; Aires *et al.* 2008; Mudge 2009)

Quantifying the error bound within the net ecosystem C balance is difficult because of a lack of independent data for verification. Baldocchi (2003) summarised studies from forest and peat bog sites with close to ideal conditions (i.e. flat topography, adequate wind) and stated that the uncertainty, with 90% confidence, under these conditions is likely to be less than ±500 kg C ha^{-1} yr^{-1}. The range of uncertainty from these reviewed studies was between ± 300 kg C ha^{-1} yr^{-1} as calculated by Goulden *et al.* (1996) for a forest ecosystem and ± 680 kg C ha^{-1} yr^{-1} calculated by Lafleur *et al.* (2001) for a peat bog. No natural grassland, semi-natural grassland or pasture sites were included in the estimate of Baldocchi (2003).

The conditions at Scott Farm can be described as close to ideal, therefore, the error bound recommended by Baldocchi (2003) has been applied to annual summed NEE and NECB for the current study. This error bound has also been applied by Mudge (2009) and Nieveen *et al.* (2005) who conducted EC studies in the same region of New Zealand. It is recommended that future studies aim to quantify error in EC measurements at this site.

3.3.5 Latent heat flux density and gross primary productivity

Latent heat flux density (λE) recorded at the EC site can be used to calculate E. Evaporation is the mass flux of water evaporated over a given area during a measured period of time, while λE is the flux density of latent heat energy stored in water vapour. Evaporation and λE are linked through the latent heat of vaporisation which defines the amount of energy required to evaporate a unit mass (1 kg) of water. GPP describes the C assimilation performed by plants as it is a measure of the total mass of C entering the ecosystem via photosynthesis. The ratio between C assimilation (GPP) and E is defined as ecosystem water use efficiency.

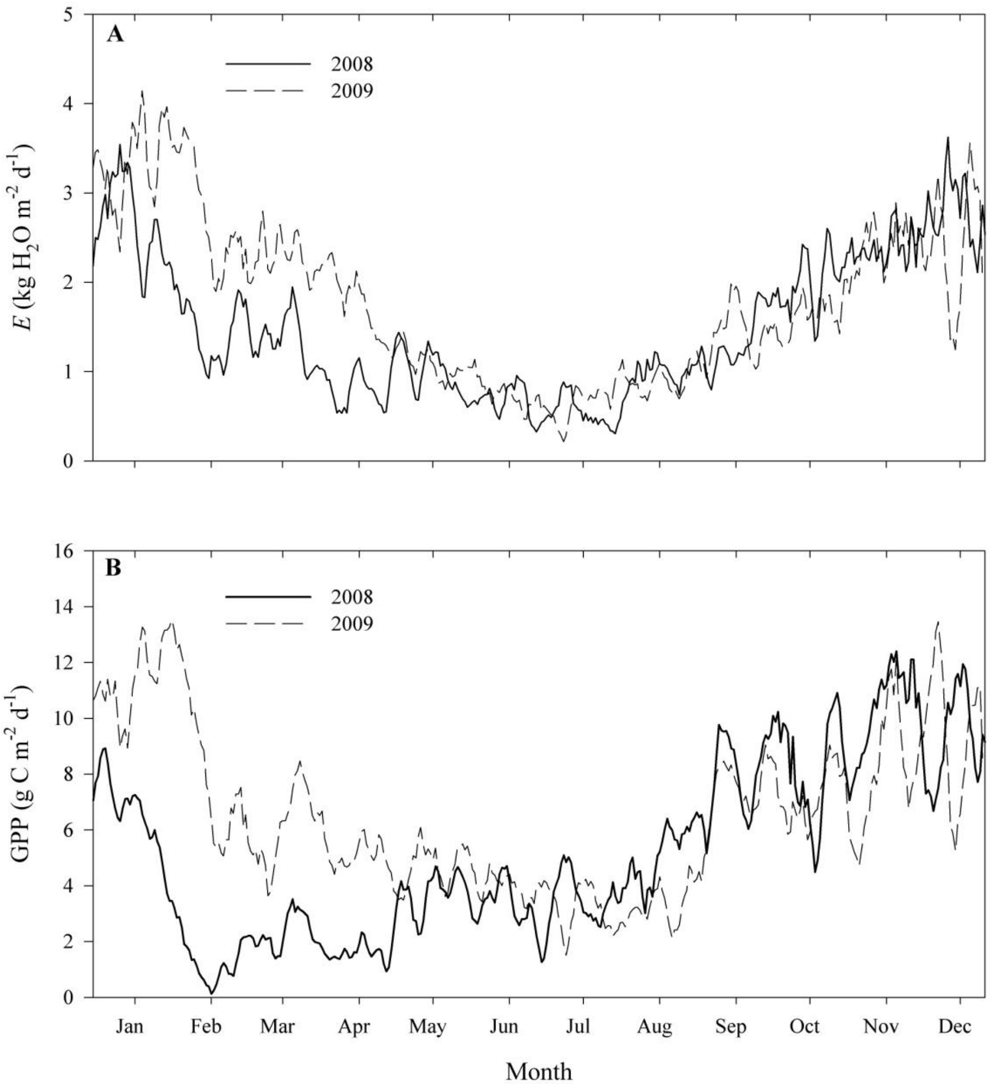

Figure 3.9 A) 5 day running mean of evaporation (*E*) and B) 5 day running mean of total gross primary productivity (GPP) for 2008 and 2009.

3.3.5.1 Evaporation and gross primary production 2008

During 2008 (Figure 3.9), GPP declined from late December through to early February when GPP practically ceased. GPP was relatively stable from April until July when GPP increased until the end of the study period (15/12/08) the short-term sharp reductions in *E* are likely in response to fluctuations in external factors such as global radiation and soil moisture content (Figure 3.2). Maximum daily GPP of 15.4 g C ha^{-1} d^{-1} was recorded on 15/11/08 while minimum daily GPP was 0.15 g C ha^{-1} d^{-1} recorded on 26/6/08.

Evaporation declined from late December and reached its annual minimum in mid June before increasing from mid July onwards. Maximum average daily *E* was 4.27 kg H_2O ha^{-1} d^{-1} which was recorded on 30/11/08 and minimum daily *E* was 0.12 kg H_2O ha^{-1} d^{-1} recorded on 30/05/08.

3.3.5.2 Evaporation and gross primary production 2009

During 2009 (Figure 3.9), GPP declined from late January until mid June before increasing for the remainder of the study from August onwards. Maximum daily GPP of 14.9 g C ha^{-1} d^{-1} was recorded on 16/01/09 while minimum daily GPP was 0.6 g C ha^{-1} d^{-1} recorded on the 26/6/09.

Evaporation declined from early January to late June before increasing from July until the end of the study (15/12/09). Maximum daily *E* of 5.07 kg H_2O ha^{-1} d^{-1} was recorded on 1/1/09 and minimum daily *E* was 0.12 kg H_2O ha^{-1} d^{-1} recorded on 26/6/09.

3.3.5.3 Inter-annual variability

GPP for 2008 practically ceased during early February due to drought, this rapid decrease produced a substantial difference in GPP between December and mid April 2008 and 2009. Following May the difference between daily GPP was not substantial and both years followed a general increasing trend until the end of the study period.

Evaporation during 2008 declined from January to mid April before becoming stable. Evaporation followed the same trend in 2009, however, there was a substantial difference in the decline measured between the two years during this period. Following May, *E* for 2008 and 2009 are not substantially different, with *E* being stable through May, June and July before increasing for the remainder of the study period.

3.3.6 Water use efficiency

Ecosystem water use efficiency (WUE g C kg^{-1} H_2O) can be calculated as the ratio between gap filled GPP (g C ha^{-1} d^{-1}) and *E* (kg H_2O ha^{-1} d^{-1}) (Fig 3.10). WUE was calculated in this study using daily totals of GPP and *E* and is plotted in Figure 3.10 as a five day running mean in order to reduce spikes found in daily WUE.

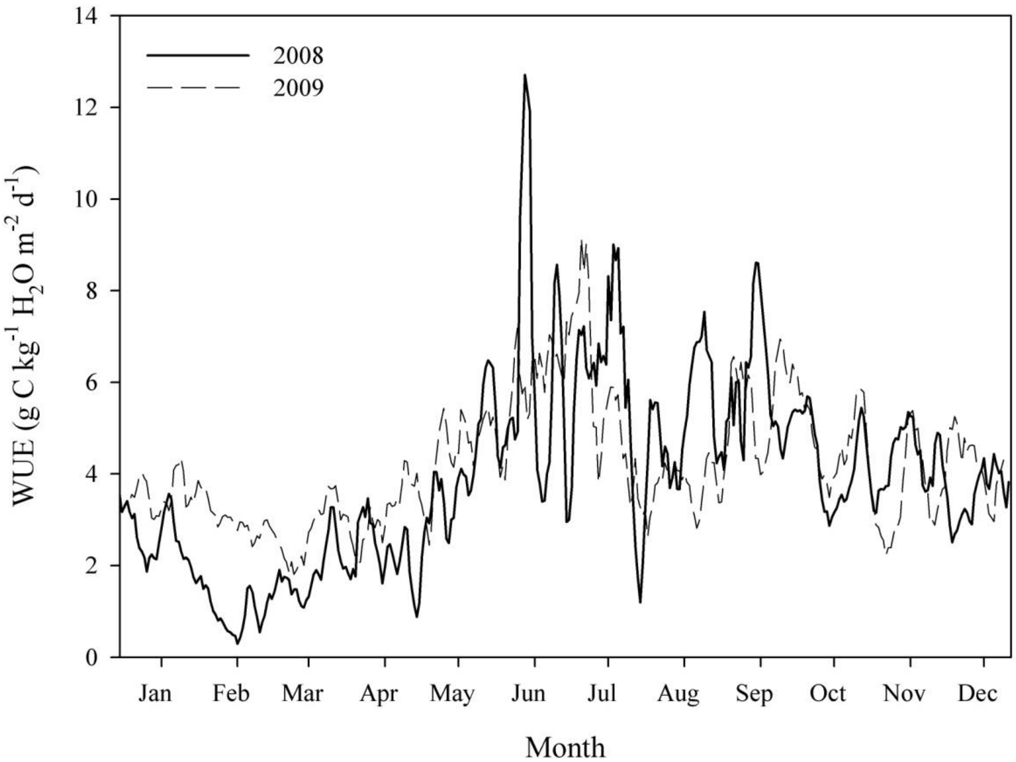

Figure 3.10 Five day running mean of ecosystem WUE at Scott Farm for 2008 and 2009.

3.3.6.1 WUE 2008

During 2008, large spikes occurred in WUE when *E* declined below 0.2 kg H_2O ha^{-1} d^{-1} and GPP remained large. Data were manually filtered by comparing spikes against various other EC and meteorological data, erroneous values were discarded.

WUE during 2008 declined from 15 December 2007 before reaching a minimum of 0.18 g C kg^{-1} H_2O d^{-1} on 11 February when GPP had almost ceased (Figure 3.10). WUE increased from April onwards reaching a maximum of 13 g C kg^{-1} H_2O on 14 June, WUE then remained relatively stable through the winter before starting to decline from September onwards. This maximum is likely a spike, however, the data was retained as it was deemed to be “real”. Normal “maximum” WUE at Scott Farm is between 6 and 8 g C kg^{-1} H_2O (Figure 3.10).

3.3.6.2 WUE 2009

WUE declined from December 2008 until late February 2009, WUE then increased from this point and reached a maximum of 11.5 g C kg^{-1} H_2O on 24 June (Figure 3.10). Following this maximum WUE declined during July and August following the cold months of May and June, where pasture production was inhibited by low soil temperature. WUE then increased during September and slowly declined from November onwards.

3.3.6.3 Inter-annual variability

Variability in WUE between 2008 and 2009 mainly occurred during summer due to the decline in GPP associated with the 2008 drought. Following summer, WUE for 2008 and 2009 follow a similar seasonal pattern with maximum WUE occurring in early winter (June – August) and minimum WUE occurring in summer (December – February).

3.3.6.4 Controls on WUE

Previous WUE studies have found that increasing vapour pressure deficit (VPD) results in a decline in WUE (Schapendonk *et al.* 1997; Smith 2003), this relationship occurs because an increase in VPD results in an increase in *E* which reduces the ratio between C gain and water loss. However, when WUE measured at Scott Farm was plotted against VPD no obvious relationship could be identified (Figure 3.11). WUE was then plotted against PPFD (Figure 3.11) which produced a slightly negative relationship (R^2 = 0.22). The observed negative relationship between WUE and PPFD is driven by the fact that at low PPFD photosynthesis (GPP) operates at its maximum rate, while *E* is minimal during times of low PPFD. This relationship causes WUE to increase when PPFD is low.

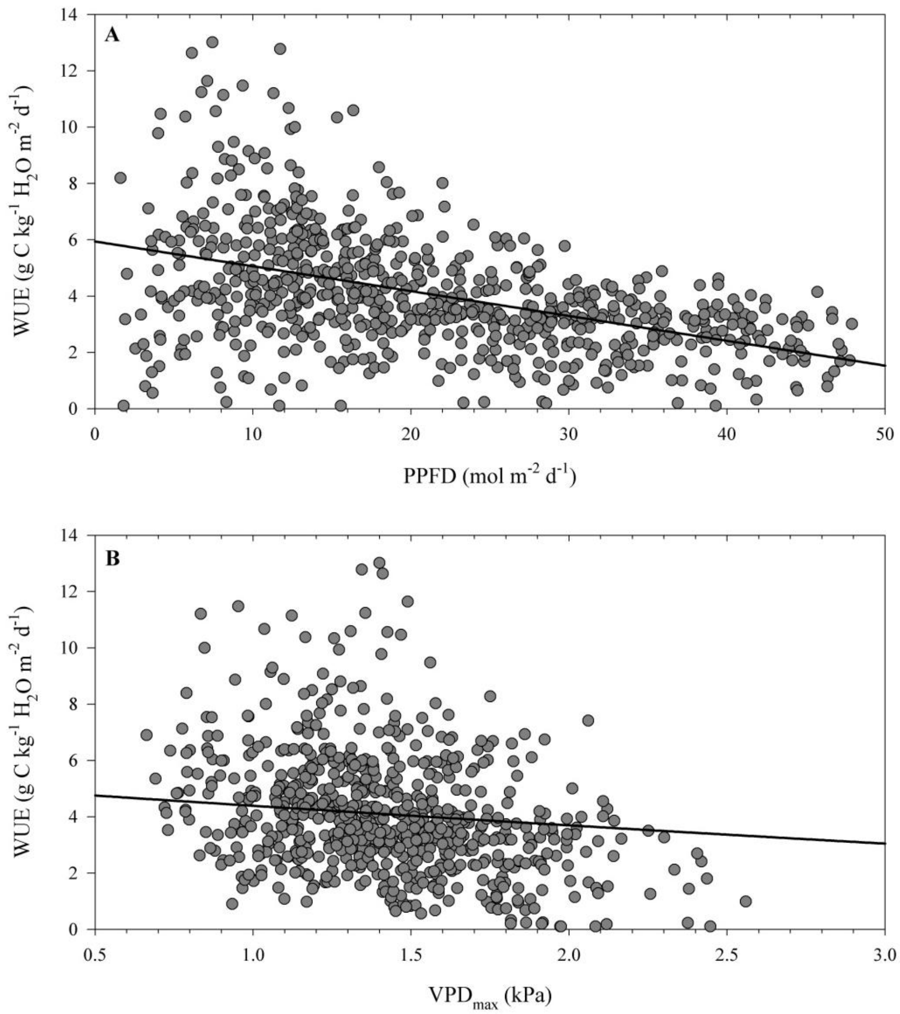

Figure 3.11 A) WUE vs PPFD, R^2 = 0.22. B) WUE vs daily max VPD, R^2 = 0.02.

3.3.7 Response of NEE to PPFD

Previous studies have found that between 80 – 95% of variation in NEE can be explained by changes in PPFD from grasslands over short time periods (Xu & Baldocchi 2004; Gilmanov *et al.* 2007; Aires *et al.* 2008). Figure 3.12 compares the response of half hourly daytime NEE to PPFD between each month over the two year study period. In 2008 Scott Farm was a greater source of C than 2009 between January and May, and NEE was less responsive to an increase in PPFD due to reduced GPP associated with the drought that reached its peak in early February. Between May and June light response of the pasture was similar between both years. July and August during 2009 were a greater source of C than the same period in 2008 due to the reduced GPP associated with the cold winter. September to December

2008 and 2009 were very similar with December 2008 and December 2009 having the most similar light response of all months.

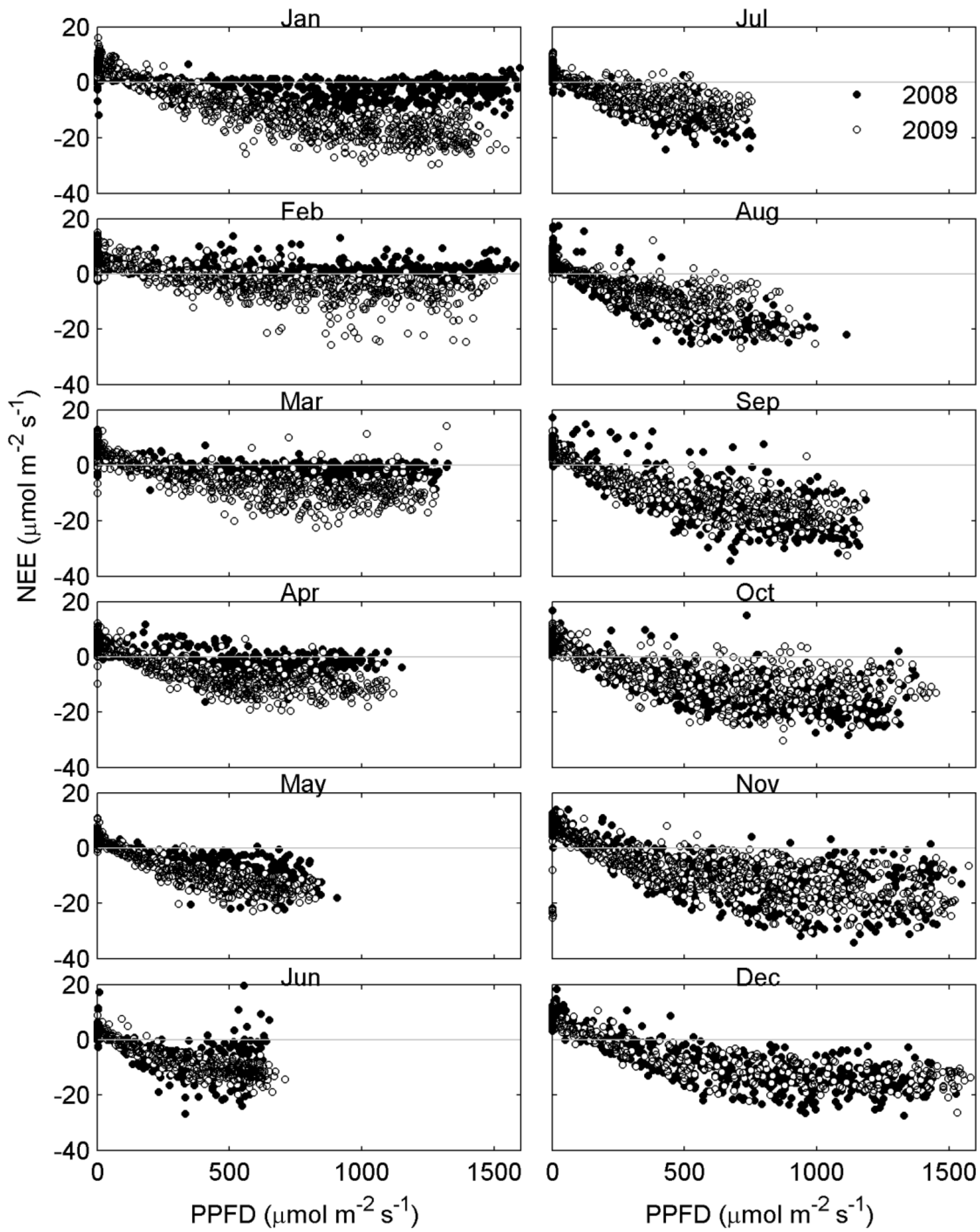

Figure 3.12 NEE vs PPFD for each month during 2008 and 2009.

3.4 Discussion

3.4.1 NEE and NECB variability

Annual cumulative NEE for 2009 (15/12/08 – 15/12/09) was -2,280 kg C ha^{-1}, with estimated annual GPP of 23,895 kg C ha^{-1} and annual TER of 21,615 kg C ha^{-1}. In comparison, annual NEE (15/12/07-15/12/08) for 2008 was -1,212 kg C ha^{-1}, GPP was 19,244 kg C ha^{-1} and TER was 18,032 kg C ha^{-1}. Inter-annual difference in NEE was 1,068 kg C ha^{-1}.

NEE recorded at Scott Farm is comparable to that recorded over a two year period by Jaksic *et al.* (2006) who measured cumulative NEE of -1,930 and -2,580 kg C ha^{-1} from dairy pasture in Ireland for 2002 and 2003 respectively. Soussana *et al.* (2007) measured NEE over a two year period from six different European grazed ecosystems. The range of NEE values presented for six different grazed ecosystems was between -130 kg C ha^{-1} and -4,640 kg C ha^{-1} and inter-annual variation in NEE was 1,121 kg C ha^{-1}, which is similar to the inter-annual variation measured at Scott Farm of 1,068 kg C ha^{-1}.

The two years of EC measurements collected from Scott Farm were influenced by different climatic conditions. 2008 experienced a drought during the summer and autumn period, followed by a productive winter and spring period. 2009 also had a dry period during summer, where soil moisture content declined below the lower limit of readily available water (42.9%), however, unlike 2008, soil moisture content did not decline below the lower limit of total available water (25%). Winter (June – August) in 2009 was the coldest for the last 30 years of records which caused the site to become a source of CO_2 during July 2009 and to only be a weak sink in August 2009, while in 2008 the site was a strong sink of CO_2 during these months. This difference was likely the result of the cold temperatures in 2009 reducing GPP which is evident in the light response plots for July and August (Figure 3.12) and from mean monthly NEE (Figure 3.4). From September to December NEE was not limited by temperature or moisture for either year (Figure 3.2) which resulted in very little inter-annual variability during this period (Figure 3.12 and 3.3). Although neither year experienced average climatic conditions, the inter-annual variation in NEE of 1,068 kg C ha^{-1} compares well to other multi-year studies (Soussana *et al.* 2007), which provides confidence that an inter-annual difference of this magnitude is to be expected.

The impact of drought on NEE has been studied previously in different ecosystems. In general drought has been observed to decrease NEE. Ammann *et al.* (2007) measured NEE of -5,170 kg C ha^{-1} during an average year and -2,150 kg C ha^{-1} during a drought year from an intensive pasture that was cut for silage quarterly. NEE of pasture grazed by sheep in Portugal during a drought year was 490 kg C ha^{-1}, while the following year which received adequate rainfall recorded NEE of -1,900 kg C ha^{-1} (Aires *et al.* 2008). NEE measured at Scott Farm during a drought year remained negative which is in contrast to the findings of Aires *et al.* (2008) but in agreement with the findings of Ammann *et al.* (2007). Xu and Baldocchi (2004) studied a grazed grassland in California and found that NEE during an average year was -1,320 kg C ha^{-1}, however, during a dry period the site became a source of C with NEE becoming 290 kg C ha^{-1}. The average decline in NEE measured from these studies is 2,340 kg C ha^{-1} which is significantly different than the difference in NEE between 2008 and 2009 measured at Scott Farm of 1,068 kg C ha^{-1}.

NEE measured at Scott Farm for 2008 was positive during summer and autumn however on an annual basis cumulative NEE was negative. The negative annual NEE measured for 2008 at Scott Farm was the result of a strong growing season following the drought, increasing GPP and hence causing NEE to decline from its initially positive state (Figure 3.4). NEE measured by Aires *et al.* (2008) did not display such a recovery, resulting in a positive annual NEE for a drought year. Differences in pasture composition and land use between ecosystems are likely to have also have influenced the difference in cumulative NEE between the two studies, the pasture composition of the paddocks at Scott Farm may have allowed for more rapid recovery following rainfall than the mixed C3/C4 grassland from the study by Aires *et al.* (2008).

NEE is the main component of the NECB calculation, however, numerous multi-year studies that have measured NEE during dry conditions do not calculate NECB for each year (Xu & Baldocchi 2004; Aires *et al.* 2008). A likely reason for this is that

C imports and exports from these non intensive ecosystems are negligible compared to the large fluxes of C that occur on an intensive dairy farm. Previous studies have demonstrated that factoring in C imports and exports to an intensively grazed ecosystem can change the final C balance significantly. For example, Nieveen *et al.* (2005) measured annual NEE of 45 kg C ha^{-1} from an intensively grazed dairy farm on a peat soil in the Waikato region, when methane production from animals and product export was accounted for, the farm became a net source of C with an NECB of 1,061 kg C ha^{-1} yr^{-1}.

When all imports and exports of C were accounted for, the NECB for Scott Farm during 2009 was -999 kg C ha^{-1} yr^{-1} (i.e. the farm was a sink of C). Using data collected by Mudge (2009), the NECB for 2008 was recalculated to be -199 kg C ha^{-1}. The NECB for Scott Farm during 2009 was similar to the average NECB measured by Soussana *et al.* (2007) from seven grazed pastures throughout Europe of -1,039 kg C ha^{-1}. Some of the grazed pastures studied by Soussana *et al.* (2007) had large C exports in the form of dry matter removal (2,200 – 4,760 kg C ha^{-1} yr^{-1}) which is much greater than in New Zealand dairy systems. NECB for Scott Farm during 2008 was less than those reported by Soussana *et al.* (2007), presumably due to the drought that occurred during later summer early autumn which substantially reduced cumulative NEE and NECB.

The difference in NECB between 2 continuous years of EC measurements from Scott Farm was 800 kg C ha^{-1}. Allard *et al.* (2007) measured the CO_2 flux from intensively grazed semi-natural grassland in France. During 2002, NECB for the intensive trial was -390 kg C ha^{-1}, 2003 was -800 kg C ha^{-1} and 2004 was -1,420 kg C ha^{-1}. This range compares well with the measured NECB from Scott Farm. Jaksic *et al.* (2006) and Byrne *et al.* (2007) calculated NECB for three years from a grazed temperate grassland in Ireland. NECB for 2002 was -240 kg C ha^{-1}, 2003 was -890 kg C ha^{-1} (Jaksic *et al.* 2006) and 2004 was -2,100 kg C ha^{-1} (Byrne *et al.* 2007). The NECB during 2004 was increased due to large feed imports (540 kg C ha^{-1}) (Byrne *et al.* 2007). Soussana *et al.* (2007) measured inter-annual variation between two years of NECB from six grazed pasture sites of 1,530 kg C ha^{-1} which is significantly greater than the variation measured at Scott Farm, this result may be due to the management practices applied on European pastures where C exports can be variable between years, whereas imports and exports at Scott Farm were relatively constant between the two years of measurements.

While NECB measured at Scott Farm for 2008 and 2009 compared well with the NECB values produced by other studies (Jaksic *et al.* 2006; Allard *et al.* 2007; Byrne *et al.* 2007) the error bound associated with measurements of NEE and NECB is large (± 500 kg C ha^{-1}). This error bound was not derived from a formal analysis and it is recommended that future studies should attempt to calculate an error bound for this study site rather than relying on estimates produced in the literature.

3.4.2 Water use efficiency

Average daily ecosystem WUE of Scott farm was 4 g C kg^{-1} H_2O and 4.2 g C kg^{-1} H_2O for 2008 and 2009 respectively (Table 3.3). Aires *et al.* (2008) measured WUE of various C3 and C4 grasses that were grazed by sheep using EC in Portugal. The average daily WUE for this ecosystem was calculated to be 3.3 g C kg^{-1} H_2O during a drought year and 4.3 g C kg^{-1} H_2O during an average year. Schapendonk *et al.* (1997) conducted a laboratory experiment involving soil monoliths (3 m^3) planted exclusively in ryegrass (*Lolium perenne*), atmospheric CO_2 concentration, air temperature and soil moisture content were regulated throughout the two year study period. WUE of the perennial ryegrass sward was 3.5 g C kg^{-1} H_2O when atmospheric CO_2 concentration was kept constant, these studies of managed pasture compare well with the average daily WUE at Scott Farm for the two year period.

Table 3.3 Comparison of WUE measured in current study against previously measured WUE of pasture, natural grassland and crops. All studies excluding Schapendonk *et al.* (1997) determined WUE using EC techniques. WUE refers to the annual average ecosystem WUE unless stated otherwise (Hunt *et al.* 2002; Law *et al.* 2002).

Reference	Location	Vegetation	Management	Year	Mean annual temp (°C)	Rain (mm yr^{-1})	WUE (g C kg^{-1} H_2O)
This study (2009)	New Zealand	Perennial ryegrass and white clover	Rotationally grazed by dairy cows	2008	14.1	1148	4
				2009	12.8	1131	4.2
Aires *et al.* (2008)	Portugal	Annual C3 and perennial C4 grasses	Grazed by sheep	2005	14.7	364	3.3
				2006	14.5	751	4.3
Hunt *et al.* (2002)[a]	New Zealand	Tussock	Grazed by sheep	2002	12.0	646	0.3
Law *et al.*(2002)[b]		Natural grassland	Natural grassland	2002			0.9
Pontoon *et al.* (2006)	Alberta, Canada	Natural grassland	Natural grassland	2005	5.7	401	1.8
Schapendonk *et al.*(1997)[c]	The Netherlands	Perennial ryegrass	Harvested	1993-1995	14.2	Irrigated	3.5

[a] Study period was 212 days
[b] Summary of growing season WUE from various FLUXNET sites, refer to www.eosdisornl.gov/FLUXNET for further information
[c] Study was conducted in laboratory environment using soil monoliths

The majority of studies that have investigated WUE have been based in forest ecosystems (Baldocchi 1997; Williams *et al.* 1998; Reichstein *et al.* 2002), making comparisons of the results from Scott Farm difficult. Law *et al.* (2002) found that average annual WUE from FLUXNET sites situated over grassland was 0.9 g C kg^{-1} H_2O with a range of 0.027 – 1.6 g C kg^{-1} H_2O. Although the mean ecosystem WUE measured at Scott Farm was greater than the mean WUE presented by Law *et al.* (2002), the ecosystem from which measurements were made must be taken into account. The FLUXNET sites summarised by Law *et al.* (2002) were natural grasslands whereas the paddocks at Scott Farm are intensively managed for pasture production, therefore, WUE is likely to be greater from pasture as production and hence C assimilation (GPP) is enhanced by species selection and fertiliser application. Law *et al.* (2002) did not report GPP from the natural grassland sites, however, Ponton *et al.* (2006) measured average annual WUE of 1.8 g C kg^{-1} H_2O from a natural grassland in Canada, where maximum GPP was 12 g C m^{-2} d^{-1} and maximum E measured was about 5 kg H_2O m^{-2} d^{-1}. Maximum GPP measured at

Scott Farm was 15.4 g C m^{-2} d^{-1} during 2008 and 14.9 g C m^{-2} d^{-1} during 2009, while maximum *E* was 4.3 kg H_2O m^{-2} d^{-1} and 5 kg H_2O m^{-2} d^{-1} for 2008 and 2009 respectively. The difference between the components of WUE from these two ecosystems is not hugely different. The difference in WUE is a product of the differences in ecosystem vegetation structure. In the natural grassland ecosystem studied by Ponton *et al.* (2006) GPP was very close to zero for four months of the year (December – February) while *E* continued through this period, resulting in very low annual average WUE. In contrast, at Scott Farm GPP was continually greater than zero (Figure 3.9) which produced a higher annual average WUE than that recorded for natural grassland.

The seasonal variation in WUE observed at Scott Farm follows a similar pattern to that observed by Aires *et al.* (2008). During both years, WUE reached its maximum in winter (June – August) and the minimum occurred in summer (December – February), with significant daily variation in ecosystem scale WUE throughout the year. Previous studies have determined that this seasonality occurs because WUE is inversely related to vapour pressure deficit (VPD) and VPD is maximum during summer and minimum during winter (Schapendonk *et al.* 1997; Law *et al.* 2002). Daily WUE was plotted against maximum daily VPD for this study (Figure 3.11), however, a significant relationship could not be identified between the two variables. WUE was then plotted against PPFD which produced a negative relationship (Figure 3.11). This relationship was also observed by Tong *et al.* (2009) who found that ecosystem WUE from maize and wheat was greater during periods of low PPFD (i.e. cloudy days). This occurs because the rate that CO_2 is fixed via photosynthesis is at its maximum at low PPFD and this rate reaches a saturation point at high PPFD where increasing PPFD has little effect on CO_2 fixation (Figure 3.12), whereas *E* continues to keep increasing at high radiation.

The two years of CO_2 and H_2O flux data collected at Scott Farm include two climatic events that decreased ecosystem WUE, the first was the drought of 2008 and the second being the cold winter of 2009. Reichstein *et al.* (2002) measured WUE during a drought from three forested ecosystems using EC and found that WUE decreased over the drought period for all three ecosystems. Aires *et al.* (2008) identified a similar relationship between drought and WUE from a grazed semi-natural grassland. Similarly, ecosystem WUE at Scott Farm declined during the drought of 2008. Although pasture and forest are different ecosystems it is interesting that both exhibit a similar WUE pattern during times of moisture deficit. Hunt *et al.* (2002) investigated WUE of tussock grassland following drought and found that WUE increased as the tussock had adapted to the moisture limitation. Although pasture did not adapt to drought conditions the ecosystem did adapt to the soil moisture deficit. During drought the main form of above ground biomass changed from ryegrass and white clover to species of weeds with tap roots that could access soil moisture stored at greater depths. WUE at Scott Farm did increase once soil moisture had recovered, however, this was because the pasture was no longer moisture stressed and unable to photosynthesise efficiently rather than an adaption of plant physiology.

Winter 2009 was the coldest in the 30 year record (NIWA 2009). Global radiation was similar to the 10 year average during winter 2009 so it is likely that the decline in air temperature rather than an increase in cloud cover forced the observed decline in GPP when E was at its yearly minimum. This combination resulted in a decline in WUE during winter. Other studies have found a similar impact of cold temperatures on WUE. Shen *et al.* (2009) measured WUE from a lucerne crop for four consecutive years, the lowest WUE recorded during this study period occurred when temperatures were the coldest and there was frequent rainfall.

3.5 Summary

Annual net ecosystem CO_2 exchange differed between 2008 (-1,212 ± 500 kg C ha^{-1}) and 2009 (-2,280 ± 500 kg C ha^{-1}) at Scott Farm. These values were comparable to international studies that applied EC over pasture (Jaksic *et al.* 2006; Soussana *et al.* 2007). The difference between annual NECB (800 kg C ha^{-1}) measured at Scott Farm was comparable with the inter-annual difference measured by some European studies (Jaksic *et al.* 2006; Allard *et al.* 2007; Byrne *et al.* 2007) but contrasted with others (Soussana *et al.* 2007) which is likely a result of variable land management practices between farm systems.

Direct comparison of the NECB estimated for 2008 with other studies is difficult because many of the studies of pasture that measured NEE during drought conditions have not calculated NECB (Xu & Baldocchi 2004; Aires *et al.* 2008), this is likely a result of C imports and exports at these sites being negligible compared to those that occur at Scott Farm. The large error bound associated with the NEE and NECB measurements made at Scott Farm is a major limitation when attempting to compare measurements with other studies or determine the annual C state of the site and it is recommended that future studies attempt to quantify this error bound.

Average daily WUE for 2008 (4.0 g C kg H_2O) and 2009 (4.2 g C kg H_2O) were similar to WUE measured at a pasture site grazed by sheep (Aires *et al.* 2008). The drought of 2008 and the cold winter during 2009 both caused declines in daily WUE, these climatic events have caused similar results in studies of forest (Reichstein *et al.* 2002), pasture (Aires *et al.* 2008) and natural grassland (Shen *et al.* 2009).

CHAPTER FOUR

Impact of cultivation on soil C

4.1 Introduction

Soil carbon (C) has a positive relationship with soil quality (Haynes 2005), increasing soil C improves soil quality which is vital for the maintenance of agricultural production from intensively grazed pasture. Soil C content of New Zealand's pasture soils has previously been assumed to be relatively stable (Jackman 1964; Tate *et al.* 2005), however, recent research has identified that New Zealand soils under pasture on flat to rolling land (commonly under dairy land use) have lost about 1 t C ha^{-1} yr^{-1} for 17 – 30 years prior to 2005 (Schipper *et al.* 2007). Similar sized losses have been identified internationally (England, Wales and Belgium) across a range of soils (Bellamy *et al.* 2005; Lettens *et al.* 2005; Meersmans *et al.* 2009), however, these losses were not exclusively from land under pasture.

A potential cause of the measured decline in soil C from New Zealand's flat to rolling pasture is the increase in cultivation frequency on dairy farms (Clark *et al.* 2007). Historically, pasture was renewed on a typical New Zealand dairy farm once every 10 – 15 years this frequency has now increased to once every 5 – 10 years (Pasture Renewal Charitable Trust 2009). Continuous cultivation is known to produce a decline in soil C content of between 20 and 60% (Davidson & Ackerman 1993; Guo & Gifford 2002) due to the reduction of C input from photosynthesis, removal of C from the system in the form of dry matter and accelerated decomposition of previously protected C due to soil aggregate destruction (Six *et al.* 2004; Grandy & Robertson 2006).

While continuous cultivation is generally limited to intensive cropping, periodic cultivation for pasture renewal and fodder crop production is becoming more common in New Zealand dairy systems (Clark *et al.* 2007). In a recent New Zealand study of periodic cultivation on a New Zealand dairy farm, Mudge (2009) measured soil C loss following a single tillage event during a drought of 1496 ± 107 kg C ha^{-1}

(mean ± standard error) from a cultivated paddock and 1446 ± 40 kg C ha^{-1} from pasture. These losses were likely not significantly different from one another because of the unusual climatic conditions that occurred during the study (100 year drought) which suppressed soil respiration.

The objective of this study was to quantify soil C loss from a Horotiu silt loam (Typic Orthic Allophanic soil) and a Te Kowhai silt loam (Typic Orthic Gley soil) (Hewitt 1998) following cultivation to determine the impact of periodic cultivation on the net ecosystem carbon balance (NECB) of a typical New Zealand dairy farm. This study was carried out during a year with more average rainfall patterns in contrast to Mudge (2009).

4.1.1 Site description

This study was conducted at DairyNZ's large-scale research site, Scott Farm (NZMS 260 S14 271E 637N), Waikato, New Zealand (Figure 4.1). The field site was a fully operational dairy farm system encompassing common field conditions and land management practices. Scott Farm was established by DairyNZ to further research on large-scale farm systems trials with one main objective being to measure the economic and environmental effects of different management practices on a dairy farm. Dairy cows were grazed on pasture outdoors year round which is a typical management practice on New Zealand dairy farms.

Scott Farm contains a number of soil types including peats, sand loams and silt loams. This study was carried out on two soils that occur in close proximity to one another; the Horotiu and Te Kowhai silt loams. Both soils were formed from the same parent material (Hinuera formation) which is a volcanogenic alluvium consisting of coarse gravels and sand deposited by the ancient Waikato River (Singleton 1991). Table 4.1 summarises the physical properties of both soils between 0 and 30 cm depth.

Carbon content of both cultivated soils was measured from 0 – 10, 10 – 20 and 20 – 30 cm depths following crop establishment (24/9/09), bulk density cores were also taken from these depths so total soil C content could be estimated for 0 – 30 cm depth (Table 4.1). Total C content for the top 30 cm of soil was estimated to be 157 t C ha^{-1}

for the allophanic Horotiu soil and 124 t C ha^{-1} for the non-allophanic Te Kowhai soil.

Table 4.1 Dry bulk density, % C and % N for the Te Kowhai and Horotiu soil.

	Te Kowhai			Horotiu		
Depth (cm)	BD* (g cm^{-3})	C (%)	N (%)	BD (g cm^{-3})	C (%)	N (%)
10	0.9	6.6	0.73	1.1	5.4	0.59
20	0.8	5.5	0.62	0.9	6.1	0.67
30	1.0	2.5	0.55	0.7	6.3	0.66

***BD= Dry bulk density.**

The Te Kowhai silt loam is a Typic Orthic Gley soil (Hewitt 1998) with a silt loam texture in the top soil and a silty clay texture in the subsoil. The primary clay mineral present in the Te Kowhai soil is halloysite which causes the soil to be poorly drained (Singleton 1991). The texture of the upper 12 cm of soil determined using a Malvern Mastersizer 2000 (Malvern Instruments Ltd, Worcestershire, UK) was 33 % sand 64 % silt and 3 % clay. Average total carbon and nitrogen between 0 and 30 cm soil depth were 4.9 % and 0.5 % respectively.

The Horotiu silt loam is a Typic Orthic Allophanic soil (Hewitt 1998) with a silt loam texture in the upper horizon and a loamy sand subsoil. The primary clay formed in the Horotiu soil is allophane which gives the soil a friable texture with a low bulk density and a high permeability. Texture of the upper 12 cm of soil determined using a Malvern Mastersizer 2000 (Malvern Instruments Ltd, Worcestershire, UK) was 30 % sand 67 % silt and 3 % clay. Average total carbon and nitrogen between 0 and 30 cm soil depth were 5.9 % and 0.64 % respectively.

Table 4.2 Vegetation sown into each of the study paddocks at Scott Farm post cultivation.

Paddock ID	Soil	Vegetation
C10	Te Kowhai	Lucerne
C10A	Te Kowhai	Lucerne
C18	Te Kowhai	Chicory & red clover
C18A	Horotiu	Chicory & red clover
C32A	Horotiu	Chicory & red clover
C36A	Horotiu	Lucerne

Six paddocks were selected for this study (Figure 4.1); three each of Horotiu soil and Te Kowhai soil. Prior to cultivation, all paddocks were under perennial ryegrass (*Lolium perene*) and white clover (*Trifolium repens*) pasture had been rotationally grazed by dairy cows (3 cows ha^{-1}) which were part of the resource efficient dairy trial (Glassey & Clark 2009). Following cultivation paddocks C10, C10A and C36A were sown with lucerne (*Medicago sativa*) while paddocks C18, C18A and C32A were sown with a mixture of chicory (*Cichorium intybus*) and red clover (*Trifolium pratense*) (Table 4.2).

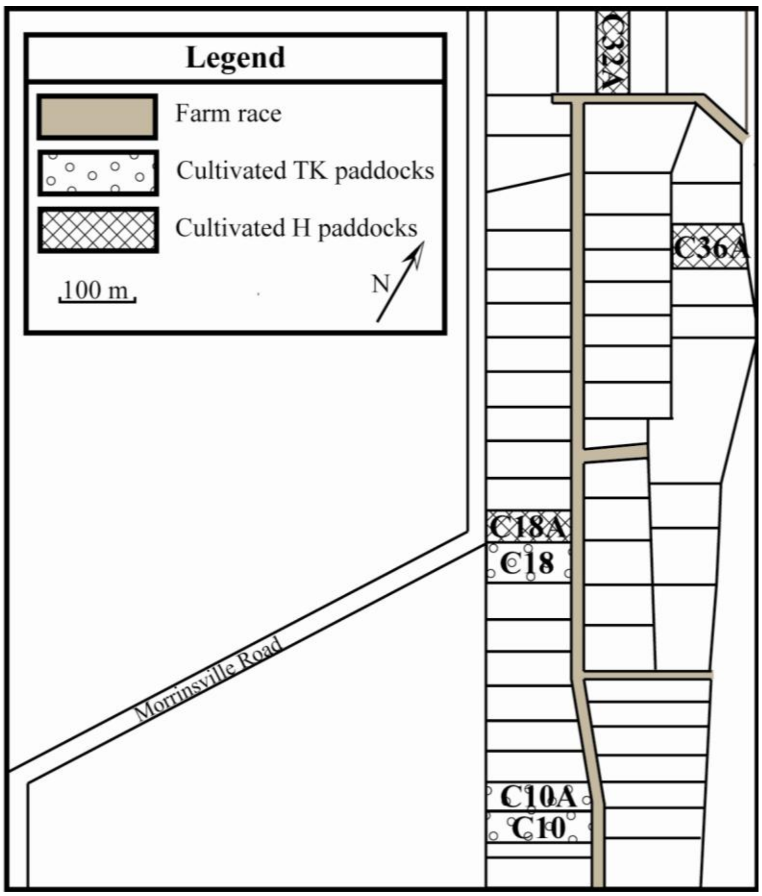

Figure 4.1 Map of Scott Farm showing the location of the Horotiu and Te Kowhai study paddocks. TK=Te Kowhai, H= Horotiu. Adapted from Scott Farm map created by DairyNZ.

4.2 Method

4.2.1 Experimental Design

This study was started on 30 September 2008 and was completed on 10 December 2008 (Table 4.3). Initially (30/09/08) measurements of CO_2 flux were taken while pasture was still present to determine if there was any significant difference in CO_2 flux between the two soils prior to cultivation. Following this period pasture was sprayed with glyphosphate-based herbicide (15/10/08) and grazed down to 1500 kg DM ha^{-1} before being incorporated into the soil using mouldboard plough to a depth of 250 mm (C Clark, pers. comm. 2008[1]). This ploughing event resulted in almost complete inversion of the top 250 mm of soil and vegetation. Power harrowing was performed following fertiliser application, resulting in the destruction of large soil clods and aggregates. Light power harrowing and spraying of Triflur based herbicide at a rate of 2 L ha^{-1} were performed prior to seed being sown with a Cambridge roller drill on 9/11/08 (C Clark, pers. comm. 2008[1]).

Measurements of CO_2 flux, temperature and moisture were not made between mouldboard ploughing and power harrowing because the large clods made it impossible to form a seal between the soil respiration chamber and the soil surface.

Table 4.3 Timeline of periodic cultivation study conducted at Scott Farm, treatments were the same for both the Horotiu and Te Kowhai soils.

Date	Day of study	Treatment	Measurements made
30/09/2008	1	Pasture	CO_2 flux, 5 cm soil temp and VWC* (every two days)
15/10/2008	16	Sprayed to kill weeds	CO_2 flux, 5 cm soil temp and VWC (every two days)
23/10/2008	24	Mouldboard plough	No measurements $^+$
28/10/2008	29	Fertilizer applied	No measurements $^+$
4/11/2008	36	Power harrow	CO_2 flux, 5 cm soil temp and VWC (daily)
9/11/2008	41	Paddocks sown	CO_2 flux, 5 cm soil temp and VWC (daily then every two days)
26/11/2008	58	Growth becomes visible	CO_2 flux, 5 cm soil temp and VWC (twice a week)

***VWC = volumetric water content, $^+$ CO_2 flux measurements not made because uneven ground limited the use of chambers.**

[1] Personal communication with Dr Cameron Clark, Scientist, DairyNZ, Hamilton.

4.2.2 Vegetation growth

Figure 4.2 shows phases of vegetation growth during the study, from initial sprayed vegetation to bare soil following power harrowing and initial germination and establishment of vegetation. Vegetation growth rate was not measured because the primary objective was to determine the loss of soil C following cultivation before pasture re-growth and root respiration had become significant.

Figure 4.2. Typical vegetation growth and soil texture during the study period.

4.2.3 Carbon dioxide emissions

Seven PVC collars (100 mm diameter by 75 mm deep) were inserted 50 mm into the top soil of each of the selected paddocks. Collar locations within the paddock were determined by first mapping soil type distribution across the paddock then applying a grid over the selected soil type which allowed seven 5 m^2 plots to be randomly selected per paddock. Grass was clipped to 25 mm to keep vegetation height uniform when measurements were initially made from pasture.

A portable infrared gas analyser (LI-8100, LI-COR Inc., Lincoln, NE, USA) and survey chamber (LI-8100-103, LI-COR Inc., Lincoln, NE, USA) were used to measure CO_2 flux from the soil collars. Initially CO_2 flux measurements were made every second day from pasture before and after spraying. Measurements commenced between 0600 and 0800 (NZST) and were completed within 3 hours. Flux measurements were made every day for the first week after power harrowing, then once every 2 days for the next three weeks. After the 26th of November 2008 measurement frequency was reduced to twice a week until completion of the trial on the 10th of December 2008 (Table 4.3).

Measurements of soil volumetric water content and temperature were made next to each collar at the same time as measurements of CO_2 flux. Volumetric water content (VWC) was measured using a Hydrosense probe (CS620, Campbell Scientific Inc., Logan, UT, USA) and was taken as the average of two adjacent measurements outside the collar. The Hydrosense was calibrated for both the Te Kowhai and Horotiu using soil collected following cultivation. Briefly, the soil was calibrated by first determining field bulk density then repacking soil to this bulk density into cores in the lab, the relationship between the gravimetric moisture content and Hydrosense moisture content was then determined for the cores and this relationship was used to correct the field measurements made by the Hydrosense (see Appendix A). Soil temperature was measured at 5 cm depth using a digital thermometer (Amadigit ad 170th, Amarell GmbH & Co., Kreuzwertheim, Germany) (± 0.05°C).

4.2.4 Data analysis

The CO_2 flux was averaged from all seven collars for every day and used to calculate a daily CO_2 flux for each paddock. Daily CO_2 flux for the Horotiu soil and Te Kowhai soil was calculated as the average of the daily CO_2 flux for each paddock.

Cumulative CO_2 loss for the Te Kowhai and Horotiu soils was determined by interpolating between the average daily CO_2 fluxes calculated for each soil and accounting for the time period between daily measurements. Uncertainty associated with daily and cumulative CO_2 flux (1 standard error, n=3) was calculated from the average daily or cumulative flux from each soil's three paddocks.

Two group t-tests were used to test whether there were any significant differences between Horotiu and Te Kowhai soils for average daily CO_2-C loss, cumulative CO_2-C loss, soil temperature or soil moisture content.

4.3 Results

4.3.1 Climate

Total rainfall measured from the nearby Ruakura climate station during the study period (30/09/08 – 10/12/08) was 411.3 mm, which was slightly greater than the 30 year average (1977 – 2008) for this period of 372 mm (NIWA 2009). Average air temperature measured at Ruakura during the study was 14.3°C which was slightly greater than the 30 year average for the period recorded at Ruakura of 13.5°C (NIWA 2009). Average 5 cm soil temperature recorded at Scott farm during the study was 17.0 °C for the Te Kowhai paddocks and 19.6 °C for the Horotiu paddocks.

4.3.2 Soil temperature

Figure 4.3 shows that throughout the study soil temperature of the Horotiu soil was significantly greater (P=0.03) than that recorded for the Te Kowhai soil. On average, the Horotiu soil was 2.5 °C warmer than the Te Kowhai soil. This difference in soil temperature was due to travel time between Horotiu and Te Kowhai paddocks as the Horotiu soils were the last to be sampled and had greater soil temperatures. The average increase in 5 cm temperature between the first and last measurement was calculated from the continuous measurements of 5 cm soil temperature of soil under pasture near the eddy covariance tower (see chapter 3, section 3.2.3). Average increase in 5 cm soil temperature was 3.6 °C, this change is likely to be responsible for the significantly greater soil temperature in the Horotiu paddocks. During the study period, no relationship between measured soil temperature and soil CO_2 flux was found.

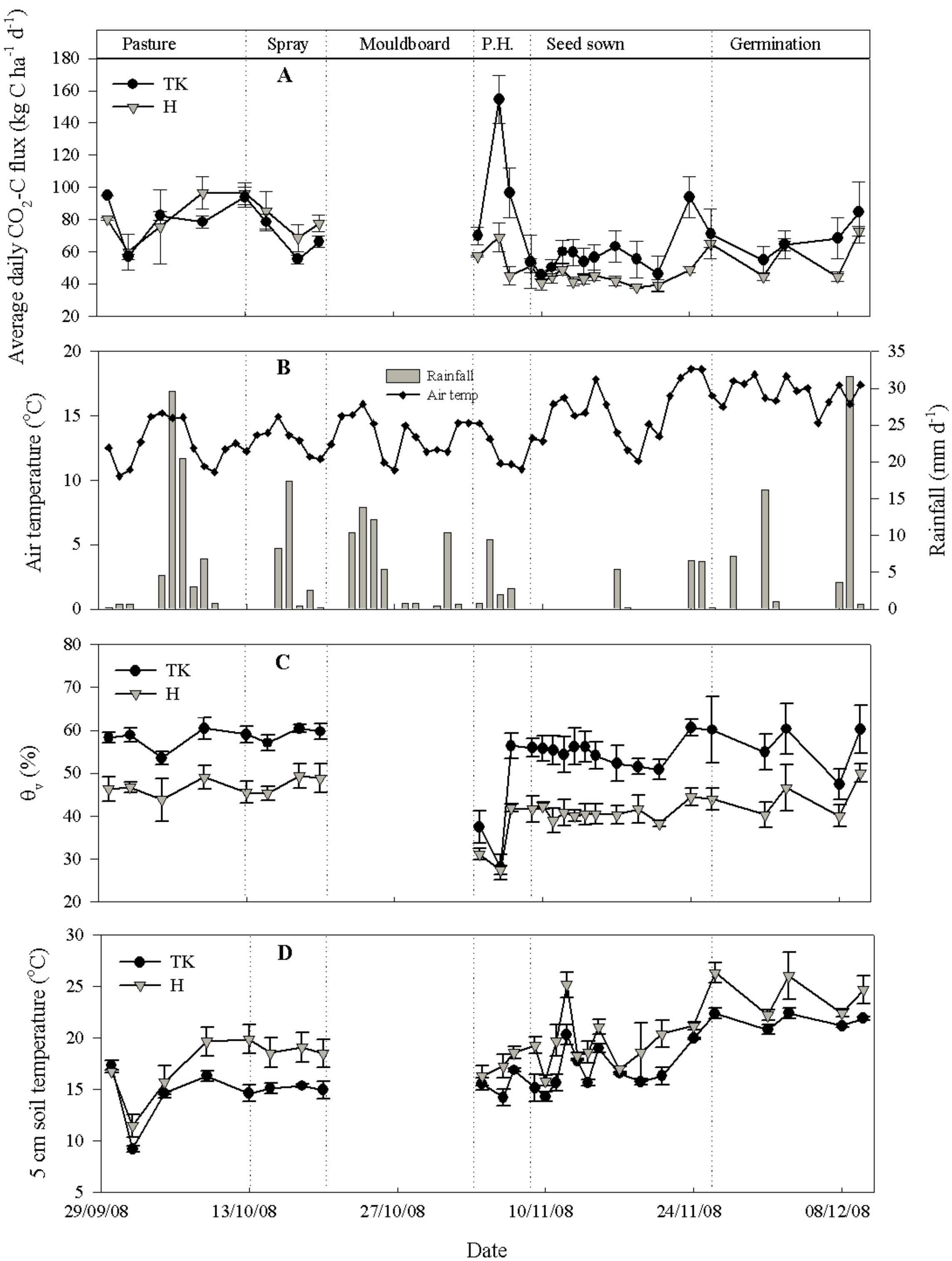

Figure 4.3 Results; (A) average daily CO_2-C flux for the Te Kowhai and Horotiu soils, (B) rainfall and air temperature measured at Scott Farm, (C) soil moisture (θ_v) and (D) 5 cm soil temperature of cultivated paddocks. P.H. = power harrow, error bars are 1 standard error (n=3).

4.3.3 Soil moisture

Soil moisture content of the Te Kowhai soil was significantly greater (P=0.04) than the Horotiu prior to cultivation. Throughout the study the Te Kowhai soil average soil moisture content was 12% greater than the Horotiu soil. Following cultivation, the moisture content of the Te Kowhai remained significantly greater (P = 0.005) than the Horotiu (Figure 4.3).

Soil moisture content of both Horotiu and Te Kowhai soils decreased following power harrowing. Rain fell on four consecutive days following power harrowing (Figure 4.3) but there was no immediate change in volumetric water content. Soil moisture content was measured to 12 cm sampling depth; moisture content at this depth was not recharged until four days after power harrowing.

4.3.4 Soil CO_2-C flux

Daily CO_2-C fluxes are displayed in Figure 4.3 and total CO_2-C losses for each management period in Table 4.4. Treatments have been consolidated into "pasture" and "cultivation", pasture is the period where autotrophic respiration was contributing to the measured flux and cultivation is the combination of CO_2 fluxes from spray, power harrow and sow when heterotrophic respiration was dominant (Table 4.5). Fluxes from the germination period were not included when estimating total CO_2 losses as autotrophic respiration may have become significant at this point.

Table 4.4 Average daily respiration and total CO_2-C losses during the study from Horotiu and Te Kowhai soils during the cultivation study.

Treatment	**Day of study**	**Average daily CO_2-C loss (kg C ha^{-1} d^{-1})**		**Total CO_2-C loss (kg C ha^{-1})**	
		Te Kowhai	Horotiu	Te Kowhai	Horotiu
Pasture	1	78.8 ± 2.0	81.3 ± 8.9	1027 ± 37	1073 ± 134
Spray	16	71.7 ± 2.8	80.2 ± 6.8	494 ± 19	554 ± 47
PH#	36	118.9 ± 10.2	57.6 ± 4.7	384 ± 35	110 ± 43
Sow	41	60.1 ± 6.7	45.2 ± 2.6	1182 ± 140	860 ± 51
Germination	60	66.5 ± 9.8	55.4 ± 2.0	921 ± 130	771 ± 27

Errors are 1 standard error (n=3), PH# is power harrow.

Table 4.5 Average daily CO_2-C flux when soils were under pasture between PH and germination.

Treatment	**Average daily CO_2-C loss ($kg\ C\ ha^{-1}\ d^{-1}$)**		**Total CO_2-C loss ($kg\ C\ ha^{-1}$)**	
	Te Kowhai	Horotiu	Te Kowhai	Horotiu
Pasture	78.8 ± 2.0	81.3 ± 8.9	1027 ± 37	1073 ± 134
Cultivation	69.0 ± 5.9	54.0 ± 4.4	2,060 ± 183	1,524 ± 141

Errors are 1 standard error (n=3).

Prior to cultivation, daily CO_2-C flux was not significantly different between the two soil types. Daily CO_2 flux from the soil under pasture was 78.8 ± 2.0 kg C ha^{-1} d^{-1} from the Te Kowhai and 81.3 ± 8.9 kg C ha^{-1} d^{-1} from the Horotiu soil (Figure 4.3). After spraying the pasture with herbicide, average CO_2 flux declined slightly to 71.7 ± 2.8 kg C ha^{-1} d^{-1} and 80.2 ± 6.8 kg C ha^{-1} d^{-1} for the Te Kowhai and Horotiu respectively. Following power harrowing, measured CO_2 flux from the Te Kowhai soil was significantly greater (P=0.04) than CO_2 flux from the Horotiu soil (Figure 4.3). Average daily CO_2 flux from the Te Kowhai was 118.9 ± 10.2 kg C ha^{-1} d^{-1} in comparison to the Horotiu which was 57.6 ± 4.7 kg C ha^{-1} d^{-1}. Fluxes decreased after seed was sown with average daily CO_2 flux of 60.1 ± 6.7 kg C ha^{-1} d^{-1} for the Te Kowhai and 45.2 ± 2.6 kg C ha^{-1} d^{-1} for the Horotiu being measured. Although fluxes decreased from both soils, the measured CO_2 flux remained significantly different (P=0.04) between the two soil types.

Average daily CO_2 flux following cultivation was 69.0 ± 5.9 kg C ha^{-1} d^{-1} for the Te Kowhai and 54.0 ± 4.4 kg C ha^{-1} d^{-1} for the Horotiu (Table 4.5). This average daily flux was not significantly different (P=0.066). However, cumulative CO_2-C loss for the cultivation period was significantly different between the two soils (P=0.047), with the Te Kowhai losing 2,059.6 ± 182.8 kg C ha^{-1} while the Horotiu lost 1,524 ± 141.5 kg C ha^{-1}.

Cumulative net ecosystem exchange (NEE) of -750 kg C ha^{-1} was measured over the ryegrass and clover pasture at Scott Farm using eddy covariance during the 41 day cultivation study period (chapter 3, section 3.2.1). NEE was partitioned into TER & GPP using the online flux partitioning model of Reichstein *et al.* (2005), TER was 3,403 kg C ha^{-1} and GPP was 4,154 kg C ha^{-1}, for a full explanation of the flux partitioning method see Appendix C.

4.4 Discussion

4.4.1 Total C loss

The average daily CO_2 loss measured following cultivation was 69 kg C ha^{-1} d^{-1} from the Te Kowhai and 54 kg C ha^{-1} d^{-1} from the Horotiu. The daily CO_2 fluxes measured in this study fall within the range of fluxes reported in recent literature (see Chapter 2, Section 2.7.1). In New Zealand, Mudge (2009) measured a CO_2 loss of 38 kg C ha^{-1} d^{-1} following cultivation of a Matangi silt loam also at DairyNZ's Scott Farm, however, the study was conducted during a drought and water limited soil respiration. In an earlier New Zealand study, Aslam *et al.* (2000) used the alkali trap method to measure a long term (341 days) CO_2 loss of 60 kg C ha^{-1} d^{-1} from a soil left fallow following cultivation. Grandy and Robertson (2006) performed a plot study where vegetation was mown before being cultivated and measurements were able to continue for 198 days, although very few measurements were made, the average daily CO_2 loss of between 54 and 43 kg C ha^{-1} d^{-1} was comparable to the present study. Yamulki & Jarvis (2002) investigated the impact of a one-off cultivation and measured CO_2 loss of 43 kg C ha^{-1} d^{-1} although the study period was short (21 days) and only six measurements were made.

Cumulative C loss was 2,059 ± 183 kg C ha^{-1} from the Te Kowhai soil and 1,524 ± 141 kg C ha^{-1} from the Horotiu soil for the 29 day measurement period, cumulative C loss was significantly different between the two soils (P=0.047). The cumulative losses of C were likely underestimated as measurements were not made during the mouldboard plough period when the soil was physically inverted and left bare for 12 days. Measurements of CO_2 loss could not be made during this time because the large clods made it impossible to form a seal between the chamber and the soil surface. Assuming soil CO_2 flux following mouldboard plough was similar to the average daily flux measured following power harrow (Te Kowhai=69.0 ± 6.4 kg C ha^{-1} d^{-1} and Horotiu = 54.0 ± 2.8 kg C $ha^{-1}d^{-1}$), cumulative soil CO_2-C loss would have been 2,880 kg C ha^{-1} from the Te Kowhai soil and 2,082 kg C ha^{-1} from the Horotiu soil had measurements been able to be made during the period between mouldboard plough and power harrow. It would be useful to be able to develop techniques to measure CO_2 losses following moldboard ploughing to confirm these estimates, however, the assumption that primary cultivation (mouldboard) and secondary cultivation (power harrow) will have a similar impact on soil CO_2 flux is supported in the literature (Reicosky & Lindstrom 1993).

Elevated soil CO_2 flux due to the cultivation event would likely have continued after vegetation started to grow and measurements were stopped. The long term recovery of soil C between periodic cultivation events is currently poorly understood (Conant *et al.* 2007). The recovery of soil C is important as it may exert greater control on the soil C store than the initial loss of C immediately following cultivation. Measuring soil CO_2 flux following vegetation establishment is problematic because measured CO_2 emission is a combination of heterotrophic and autotrophic respiration. The partitioning of autotrophic and heterotrophic respiration was attempted using the root mass regression approach following the conclusion of the cultivation trial (see Appendix B). Briefly, this technique involved measuring CO_2 flux from the soil collars multiple times on the same day before pushing the collars flush with the soil surface, collars were then excavated and taken to the laboratory. Soil from these collars was washed away and roots were retained, the roots from each collar were then dried and weighed. Root mass for the collar was plotted against total soil respiration measured from the collar and a simple linear regression was performed, by reading off the Y intercept (CO_2 flux at 0 g root mass) from this plot the contribution of root respiration to total soil respiration can be determined. The contribution of autotrophic respiration to total soil respiration ranged from 48% to 23% using this method. This approach showed promise; however, time constraints did not allow its effectiveness to be fully investigated.

4.4.2 Completing the C balance following cultivation

Measurement of CO_2 emissions following cultivation provides a conservative estimate of the true C balance. During site preparation and cultivation, photosynthesis ceased and this lack of C input needs to be accounted for to obtain total C loss for the ecosystem. Net ecosystem CO_2 exchange was calculated from the nearby eddy covariance system (see chapter 3, section 3.2.1) to calculate the potential input of C (via photosynthesis) to the cultivated paddocks had they not been cultivated.

During the 41 day cultivation period, nearby pasture was a sink of -750 kg C ha^{-1}. Assuming that the cultivated paddocks would have fixed the same amount of CO_2 as the paddocks in the eddy covariance footprint had they been in pasture then the net C loss due to cultivation can be calculated (measured CO_2 flux + NEE). The range of net soil C loss due to cultivation for the non-allophanic Te Kowhai soil was between 2,880 kg C ha^{-1} (CO_2 flux only) at its minimum and 3,630 kg C ha^{-1} (CO_2 flux + NEE) at its maximum. The range for the allophanic Horotiu soil was between a minimum of 2,082 kg C ha^{-1} (CO_2 flux only) and a maximum of 2,832 kg C ha^{-1} (CO_2 flux + NEE).

The measurements of C loss during cultivation at Scott Farm can be extrapolated to assess the impact of periodic cultivation on a typical New Zealand dairy farm with a productive area of 120 ha. It has been recommended that on an intensive dairy farm 10% of pasture should be renewed per year to maintain optimum production (Pasture Renewal Charitable Trust 2009). On a typical dairy farm (120 ha) this equates to cultivating 12 ha yr^{-1}, and if the C loss due to cultivation is assumed to be 3,231 kg C ha^{-1} (the average of the loss from Horotiu and Te Kowhai paddocks), annual cultivation on a dairy farm would result in a loss of 38.7 t C yr^{-1}. Using the NECB at Scott Farm of -999 (± 500) kg C ha^{-1} yr^{-1} for non-cultivated pasture area (108 ha) and combined with the NECB from cultivated paddocks when under pasture (890 kg C ha^{-1} yr^{-1} for 12 ha), the average dairy farm would gain about 118 t C yr^{-1}. Subtracting the maximum net loss from cultivation (39 t C yr^{-1}) from the gain by pasture (118 t C yr^{-1}) results in an overall gain of C to the 120 ha farm of 79 t C yr^{-1} or 0.66 t C ha^{-1} yr^{-1}. This C gain is a preliminary estimate, however, it does provide insight into the potential impact of cultivation on a dairy farm's net C balance and suggests that periodic cultivation is not exclusively accountable for the loss of 1 t C ha^{-1} yr^{-1} from intensively grazed flat to rolling pastures measured by Schipper *et al.* (2007).

If the same calculation is performed using the NECB for the 2007-2008 year (-199 ± 500 kg C ha^{-1}) as a total C gain to the farms area of 23,617 kg C yr^{-1} (108 ha times -199 kg C ha^{-1} plus 12 ha times 176 kg C ha^{-1}). Carbon loss due to cultivation would be equivalent to 2009, 12 ha times an average loss of 3,231 kg C ha^{-1} making C loss from cultivation 38,772 kg C yr^{-1}. This would make the dairy farm a net source of 15.4 t C yr^{-1} or 0.13 t C ha^{-1} yr^{-1}.

This loss of C has the potential to increase if dry matter was harvested and exported, following the assumptions made in Chapter 2, Section 2.5.2.3, if two harvesting events occurred from each cultivated paddock soil the farm would become a source of 0.73 t C ha^{-1} yr^{-1}. From this estimate it can be concluded that periodic cultivation is not exclusively responsible for measure declines in soil C from intensively grazed dairy farm. A combination of external factors are likely to be responsible measured soil C loss from intensively grazed dairy systems, the estimate made in this section demonstrates that the combination of drought and cultivation could result in a small loss of soil C, however, if harvest and dry matter export is included in this estimate then soil C loss could become significant.

4.4.3 Possible reasons for differences between Te Kowhai and Horotiu

The average daily CO_2 loss measured following cultivation was 69 kg C ha^{-1} d^{-1} from the Te Kowhai and 54 kg C ha^{-1} d^{-1} from the Horotiu. The range of net soil C loss due to cultivation for the non-allophanic Te Kowhai soil was between 2,880 kg C ha^{-1} (CO_2 flux only) at its minimum and 3,630 kg C ha^{-1} (CO_2 flux + NEE) at its maximum. The range for the allophanic Horotiu soil was between a minimum of 2,082 kg C ha^{-1} (CO_2 flux only) and a maximum of 2,832 kg C ha^{-1} (CO_2 flux + NEE). There are a number of possible reasons for the differences in daily and total CO_2 losses between the Te Kowhai and Horotiu soils, however, the main difference are contrasting clay mineralogy and drainage.

4.4.3.1 Clay mineralogy

The Te Kowhai and Horotiu soils have contrasting clay mineralogy. The primary clay mineral in the Te Kowhai soil is halloysite while the Horotiu soil is dominated by the clay mineral allophane (Singleton 1991). Previous research has found that allophanic soils have the greatest amount of organic C of all the mineral soil orders (Dahlgren *et al.* 2004) as allophanic clays protect soil C from decomposition (Parfitt 2009). A recent review concluded that C loss following disturbance from non-allophanic soils was greater than those occurring from allophanic soils such as the Horotiu (Parfitt 2009). The results from the current study agree with this finding as the Te Kowhai lost significantly more soil C than the allophanic Horotiu soil following cultivation. Soil C loss can also be presented as a percentage of C within the plough layer (0-30 cm), the Te Kowhai soil lost 2.3% of C within the plough layer while the Horotiu lost 1.4%.

Previous studies that measured total C stocks rather than CO_2 flux have drawn similar conclusions. In New Zealand, Parfitt *et al.* (1997) found that after 20 years of continuous cropping, total C decreased by only 10 t C ha^{-1} in an allophanic soil whereas an adjacent non-allophanic recent soil lost 23 t C ha^{-1}. Further evidence for the difference between allophanic and non-allophanic soils was presented by Parfitt *et al.* (2002) who compared C turnover rates from adjacent Horotiu and Te Kowhai soils under continuous cropping. They found that the proportion of old pasture C remaining in the soil after 25 years of continuous maize cropping was about 78% in the allophanic Horotiu soil and about 69% in the non-allophanic Te Kowhai soil (Parfitt *et al.* 2002).

Allophanic soils are thought to stabilise soil organic C because they contain high concentrations of available aluminium (Al) and allophane clay which has a high specific surface area (Percival *et al.* 2000; Matus *et al.* 2008) which increases C protection and reduces the decomposition rate of old C (Parfitt *et al.* 2002). Percival *et al.* (2000) demonstrated that Al_{py}, Fe oxide, allophane and clay concentration explained the greatest amount of variation in soil C concentration, while Al_{py} and allophane content explained the greatest amount of variation in soil C content within different soil types (Percival *et al.* 2000).

4.4.3.2 Drainage

As a result of landscape position and soil forming factors the Te Kowhai is poorly drained while the Horotiu is well drained (Singleton 1991). Previous research has shown that aerating poorly-drained soil through physical disturbance results in a larger CO_2 flux than physical disturbance of well-drained soil (Reicosky 1997; Meersmans *et al.* 2009). Cultivating poorly-drained soil results in increased aeration and increases in oxygen availability which stimulates microbial decomposition of labile C (Reicosky 1995).

Reicosky (1997) studied four different soil types under long-term (>80 years) continuous cultivation and found that the rate of CO_2 loss following cultivation was partially dependent on soil position within the landscape and that soils with very poor drainage had significantly greater CO_2 loss following cultivation than well-drained soil. Meersmans (2009) studied agricultural soils in the Flanders district of Northern Belgium and showed that soil C loss between 1960 and 2006 was strongly influenced by drainage and landscape position. The study found that moderately-poorly drained sand soils under grassland lost approximately 237 kg C ha^{-1} yr^{-1} and very-poorly drained sand soil lost approximately 870 kg C $ha^{-1}yr^{-1}$. In contrast, excessively-well drained grassland soils gained 140 kg C ha^{-1} yr^{-1} while moderately well drained soils gained 510 kg C ha^{-1} yr^{-1}.

The tendency of poorly-drained soils to lose more soil C following cultivation may have contributed to the significantly greater C loss from the poorly drained Te Kowhai soil than the well drained Horotiu.

4.4.4 Impact of moisture and temperature on CO_2 flux

Following cultivation, CO_2 flux from the Te Kowhai and Horotiu soils was inhibited by low soil moisture content following cultivation, however, no significant relationship between soil temperature and CO_2 flux was identified.

4.4.4.1 Soil moisture

Soil moisture content of the Te Kowhai soil was significantly greater ($P = 0.04$) than the Horotiu prior to cultivation. Following cultivation, the moisture content of the Te

Kowhai soil remained significantly greater than the Horotiu soil (P = 0.005) (Figure 4.3). Numerous studies have tried to describe the relationship between soil water content and respiration, with the one main conclusion being that low water contents can inhibit CO_2 production in soils (Davidson *et al.* 2000).

Following power harrowing, the soils dried rapidly, thus inhibiting soil respiration (Figure 4.3). Initial rainfall following cultivation produced a rapid increase in CO_2 flux from the dry aerated soil as an accumulation of dry labile C was rapidly decomposed (Figure 4.3). The rapid decomposition and release of CO_2 following the rewetting of a dry soil is known as the “Birch effect”, after the work of Birch who found that the rate of soil organic matter decomposition increased following the rewetting of dry soil and that this enhanced decomposition declined as time increased from the initial rewetting event (Jarvis *et al.* 2007).

4.4.4.2 Soil temperature

This study did not identify a positive relationship between increasing soil temperature and soil respiration as might be expected (Davidson & Janssens 2006). Figure 4.3 shows that throughout the study, soil temperature of the Horotiu soil was significantly greater (P=0.03) than for the Te Kowhai soil. This was primarily because the Horotiu paddocks were measured later in the day than in the Te Kowhai paddocks. However, CO_2 fluxes were consistently greater from the Te Kowhai soils despite lower temperatures suggesting differences in soil temperatures were not a major driver of differences in CO_2 losses between soils.

In addition to differences in sampling time, soil colour and moisture content may have also contributed to the significantly greater temperature from the Horotiu soil. The plough layer of the Horotiu consisted of dark topsoil extending to approximately 30 cm depth, this topsoil was well drained and the soil moisture content was significantly lower than the Te Kowhai prior to cultivation. The Te Kowhai soil had shallow dark topsoil (~20cm) and light grey subsoil which was incorporated into the plough layer. The greater moisture content of the Te Kowhai combined with the difference in colour has the potential to increase albedo of the Te Kowhai soil in comparison to the Horotiu soil and reduce 5 cm soil temperature.

4.5 Summary

Net soil C loss due to the one-off cultivation event at Scott Farm was between 2,880 kg C ha^{-1} and 3,742 kg C ha^{-1} for the non-allophanic Te Kowhai soil while the allophanic Horotiu soil lost between 2,082 kg C ha^{-1} and 2,944 kg C ha^{-1}. This range was calculated from using either CO_2 loss measured from soil (minimum loss) or CO_2 loss with NEE added (maximum loss). The soil C loss from the allophanic Horotiu soil was significantly less than that from the non-allophanic Te Kowhai soil. This finding was in agreement with previous research which has concluded that allophanic soils protect soil C from decomposition (Parfitt 2009) and that poorly drained soils are more susceptible to elevated levels of soil C decomposition following cultivation (Reicosky 1995; Reicosky 1997).

Periodic cultivation of intensively grazed dairy soils may have contributed to the decline in soil C from flat to rolling pasture measured by Schipper *et al.* (2007), however, periodic cultivation of dairy pasture is not sufficiently large to be the sole driver of the measured decline. As frequency of periodic cultivation increases, the contribution of this practice to losses of soil C is also likely to increase.

To gain further understanding about the loss of soil C associated with periodic cultivation on New Zealand dairy farms, it is recommended that an appropriate flux partitioning technique is applied. This would allow heterotrophic respiration to be estimated independent of autotrophic respiration, and allow measurements of CO_2 flux to continue after the crop or pasture has germinated.

Measuring large-scale long-term net ecosystem CO_2 exchange over a pasture renewal or cropping event using eddy covariance is also recommended as this would enable the recovery of soil C between cultivation events to be determined.

CHAPTER FIVE

Summary and conclusions

5.1 Introduction

Maintenance of soil C content is important as a relatively small percentage change in the global soil C store has the potential to cause a large change in atmospheric CO_2 concentration. Losses of soil C can also lead to a decline in soil quality and the soils capacity to be productive and carry out other services such as filtering of pollutants.

Globally, research on soil C dynamics has largely focused on forests, croplands and natural grasslands, while intensively grazed pastures have received much less attention. In New Zealand, the dynamics of soil C content and C cycling in intensively grazed dairy systems are poorly understood. A recent study has measured a long term (17-30 years) loss of about 1 t C ha^{-1} yr^{-1} from New Zealand's flat to rolling dairy pasture (Schipper *et al.* 2007).

The aim of this study has been to develop a carbon balance for 2009 and to determine the water use efficiency (WUE) of pasture for 2008 and 2009. To achieve this aim two years of eddy covariance (EC) and meteorological data were analysed to determine the variability of net ecosystem CO_2 exchange (NEE), WUE and net ecosystem carbon balance (NECB) at DairyNZ's Scott Farm, Waikato. The impact of periodic cultivation on the C balance was also assessed by measuring the CO_2 loss associated with a single spring cultivation event on adjacent allophanic and non-allophanic soils using the closed chamber technique.

5.2 Annual carbon balance

The NECB measured at Scott Farm for 2008 was -199 ± 500 kg C ha^{-1} while the C balance for 2009 was -1,014 ± 500 kg C ha^{-1}, these values compare well with the average NECB of -1,039 kg C ha^{-1} yr^{-1} for a range of European grasslands (Soussana *et al.* 2007). Annual NEE was substantially different between 2008 (-1,212 ± 500 kg C ha^{-1}) and 2009 (-2,280 ± 500 kg C ha^{-1}) at Scott Farm, the difference between years of 1,068 kg C ha^{-1} agreed with the range measured in studies of grazed pasture over multiple years (Jaksic *et al.* 2006; Soussana *et al.* 2007). It is difficult to calculate error bounds for EC data so the error bound applied in this study is simply an estimate of the uncertainty at 90% confidence, this estimate was derived from previous studies that performed EC at sites with close to ideal conditions (Goulden *et al.* 1996; Lafleur *et al.* 2001).

The difference between NEE and NECB in 2008 highlighted the importance of calculating NECB annually as NEE during 2008 suggested the system was a sink of C, whereas, once C imports and exports were accounted the C balance was not significantly different from zero.

The site was not an annual source of C in 2008 or 2009, which is in contrast to the findings of Schipper *et al.*(2007) who measured a long term decline in soil C of 1 t C ha^{-1} yr^{-1} over 17 – 30 years. However, in shorter time frames, pastures can oscillate between sinks and sources of C depending on changes in external factors such as climate (Schipper *et al.* in press). EC measurements at this site need to be continued before conclusions about long term changes in C can be drawn.

5.3 Water use efficiency

WUE is calculated as the ratio between C assimilation (measured as gross primary production (GPP)) and evaporation (calculated from measured latent heat flux density (λE)), this ratio is an important indicator of pasture survival, productivity and fitness (Ponton *et al.* 2006). WUE is also important for helping to understand how potential changes in climate will impact the C and energy budgets of grazed pasture (Ponton *et al.* 2006).

Average daily WUE for 2008 (4 g C kg^{-1} H_2O) and 2009 (4.2 g C kg^{-1} H_2O) were not substantially different from one another and compared well to international field and laboratory studies of pasture (Schapendonk *et al.* 1997; Aires *et al.* 2008). Overall, WUE displayed a seasonal pattern with a minimum in the summer (December – February) and maximum in the winter (June – August). This seasonal pattern agreed with that measured by Aires *et al.* (2008) for a mixed pasture site grazed by sheep.

During the drought of 2008, WUE declined which is in agreement with studies of forested (Reichstein *et al.* 2002) and grazed pasture (Aires *et al.* 2008) ecosystems that experienced drought. Winter 2009 caused a decline in WUE, this deviation from regular seasonality was caused by the 2009 having the coldest winter in the last 30 years, as has also been observed in natural grassland (Shen *et al.* 2009).

WUE has not previously been measured for a typical New Zealand dairy pasture by EC and the results from this study demonstrate that this ecosystem experiences greater productivity than semi-natural and natural grassland, and that ecosystem WUE is sensitive to annual climatic variation.

5.4 Carbon loss following cultivation

Soil CO_2-C loss following cultivation was measured using the closed chamber technique. During cultivation photosynthesis ceased as all plant cover was removed. Potential C input (NEE) to pasture during the cultivation period was -750 kg C ha^{-1} at the adjacent EC site. To calculate the maximum net soil CO_2-C loss, this potential C input must be added to measured CO_2-C emissions. Soil C loss from the Te Kowhai soil was between 2,880 kg C ha^{-1} (CO_2-C flux only) and 3,742 kg C ha^{-1} (CO_2-C flux + NEE) while the Horotiu soil lost between 2,082 kg C ha^{-1} (CO_2-C flux only) and 2,944 kg C ha^{-1} (CO_2-C flux + NEE). The soil C loss from the allophanic Horotiu soil was significantly less than that from the non-allophanic Te Kowhai soil. This result is likely due to allophanic clays found in the Horotiu protecting soil C from decomposition (Parfitt 2009). Additionally, poorly drained soils such as the Te Kowhai tend to lose more C following cultivation due to aeration caused by cultivation which increases oxygen penetration into the soil and accelerates decomposition of soil C (Reicosky 1995). The Te Kowhai soil has been previously shown to lose significantly more soil C than the Horotiu soil in a long term (>20 years) study of continuous cultivation (Parfitt *et al.* 1997; Parfitt *et al.* 2002).

When the C loss associated with cultivation is included in a net calculation of annual farm scale C balance, it can be concluded that periodic cultivation of intensively grazed dairy soils cannot be exclusively responsible for the measured decline in soil C content of 1 t C ha^{-1} yr^{-1} from flat to rolling pasture as measured by Schipper *et al.*

(2007). However, increasing cultivation frequency may increase C loss from pasture soils if soil C does not fully recover between cultivation events. Furthermore, it is currently unclear how much soil C recovers between cultivation events. Long term soil C recovery rate is likely to be equally as important for the soil C content of an intensively grazed dairy soil as the short term CO_2-C loss measured in this study.

To understand the recovery of soil C following cultivation, it is recommended that EC is used to measure CO_2 exchange following a cultivation event and subsequent plant growth, to measure the net ecosystem CO_2 exchange including the long term effect on a dairy system's C balance.

5.5 Further research

The continuation of EC measurements at Scott Farm is recommended because the conclusion that this intensively grazed pasture is a net sink of C needs to be supported by further CO_2 exchange measurements coupled with measurements and estimates of imports and exports of C.

The use of EC to measure CO_2 exchange through the complete cultivation and recovery cycle of a cropping or pastoral renewal event would also be beneficial because the amount of C recovered between cultivation events is currently unknown and this recovery rate could contribute to long term soil C content change in dairy systems.

Along with periodic cultivation there are numerous other land management practices that are currently increasing in New Zealand such as strip grazing, which could be effectively studied using advanced chamber techniques. The effects of these management practices of C exchange are poorly understood.

It is recommended that the EC approach for measuring C dynamics be compared to the direct measurement of soil C to determine what a negative NECB (i.e. a C sink) really means for the soil C store of an intensive dairy system. In theory, if a soil was to record an average NECB of -1 t C ha^{-1} for ten years then direct measurement of soil C between the first year and the final year should identify a gain of approximately 1 t of C ha^{-1} yr^{-1}. The problem with testing this hypothesis is that long term EC measurements and direct soil C measurements are required for the same site, and these data are currently un-available as the two measurement techniques are rarely used in unison.

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APPENDIX A

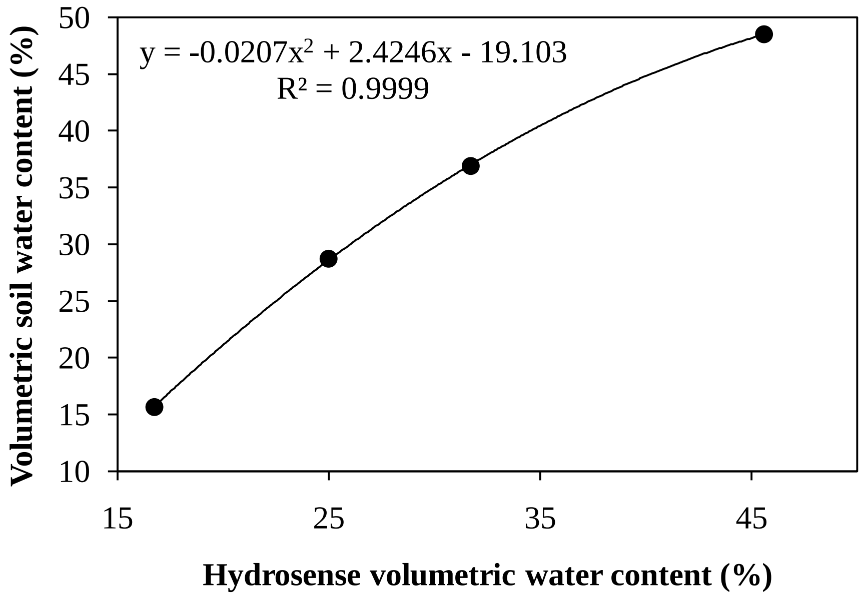
Instrument calibration

The CS620 Hydrosense probe (Campbell Scientific Inc, Logan, UT) was calibrated for the Te Kowhai and Horotiu soil during August 2009.

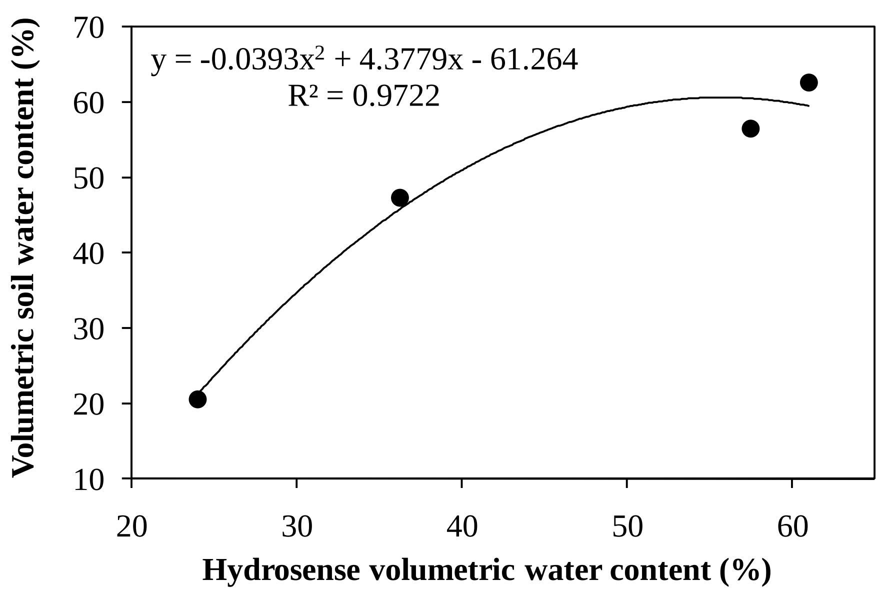
Two soil cores (98 mm diameter by 75 mm deep) were collected from each of the cultivation study paddocks. The cores were taken from 0-75 mm depth then dry bulk density was determined by drying the cores at 105°C for 48 hours and then weighing.

Additional Te Kowhai and Horotiu soil was collected from 0 – 120 mm depth at the same time as bulk density cores were taken. Soil was sieved on return to the laboratory, to homogenise soils. The soil was then separated into four equal sized samples. One of these samples was retained at field moist state while another had water added and the two other samples were air dried for 24 and 48 hours to produce four different moisture contents. Each soil sample was then packed into a PVC cylinder (105 mm in diameter by 250 mm deep) and compacted to the same bulk density as that measured in the field.

Average volumetric moisture content was measured for each soil using standard laboratory procedure. A sub sample of loose field moist soil (~ 5 g) was collected, weighed and dried at 105 °C for 48 hours. The soil was reweighed and gravimetric water content calculated. Soil moisture measured in the lab was then plotted against the average of 5 measurements of volumetric water content taken using the Hydrosense probe inserted into the repacked soils (Figure A.1 & Figure A.2). The relationship between the two measurements allowed field measurements to be calibrated to laboratory results.



A.1 Calibration curve and equation for Horotiu soil.



B.2 Calibration curve and equation for Te Kowhai soil.

APPENDIX B

Root mass regression

Introduction

The partitioning of the components of ecosystem respiration into autotrophic and heterotrophic sources is an area of C cycling which requires a simple, effective method to be used universally throughout multiple ecosystems. Kuzyakov (2006) reviewed multiple approaches and concluded that the root mass regression approach could be applied to numerous ecosystems. The regression approach has two main limitations; the method assumes that root and microbial respiration rates are equally affected by temperature and it assumes that microbial respiration is independent of root biomass.

Method

The root mass regression method was used to partition the CO_2 flux measured by the closed chamber technique from soil collars at the Scott farm trial. First, respiration was measured from each soil collar in two sampling rounds (2 measurements per collar) between 0500 and 0800 at Scott Farm (19th Dec). Collars were then pushed flush with the soil surface, excavated, wrapped and labelled. The soil cores were then transferred to HortResearch, Palmerston North where a pneumatic root washer was used to separate the soil and root components of the soil. The pneumatic root washer pumped water and air over a stack of sieves (1 mm & 0.5 mm sieve) which contained soil, and water and soil moved through the sieve while roots were retained (see digital Appendix E for photographs). Roots were recovered by floating sieves in water.

Due to time restraints all roots were collected but roots were not seperated into dead and live groups. Due to the very small diameter of fine roots, not all root biomass was retained on the sieves. Roots were washed thoroughly, however, clay particles were still retained which would have increased the dry mass of the roots and modified the final regression result. Collected roots were then transferred to marked paper bags and dried at 70°C for 24 hours. Once dry, roots were weighed to allow root mass regression to be performed. To improve the method applied by this study it is recommended that soil samples are soaked for 24 hours prior to sieving and that calgon is used to disperse clays from root biomass.

Results

Figures B.1 and B.2 display regressions from the Te Kowhai soil, some individual paddocks produced good regressions, however results from each soil were analysed as a whole (Fig. B.2) there was only weak or no correlation between root mass and respiration rate.

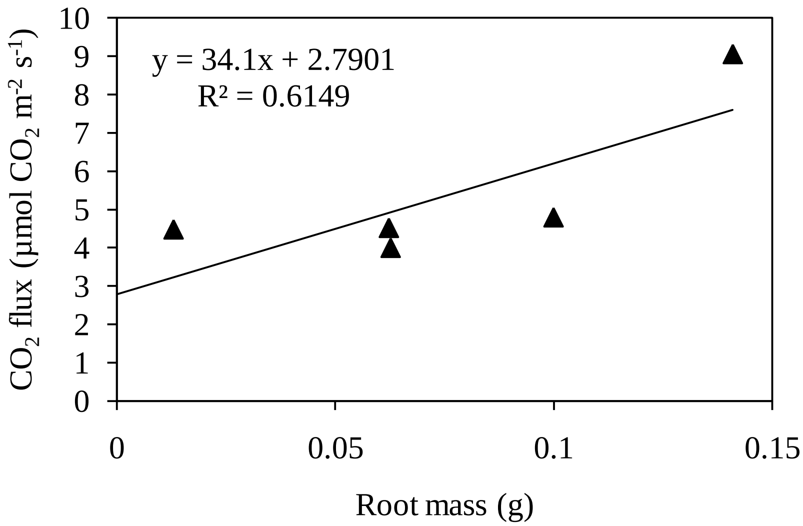


Figure B.1. Root mass regression plot for a single Te Kowhai paddock. Hetrotrophic respiration which is determined from Y axis intercept (i.e. zero root mass) contributed 52% to average respiration for the paddock of 5.4 µmol CO_2 m^{-2} s^{-1}.

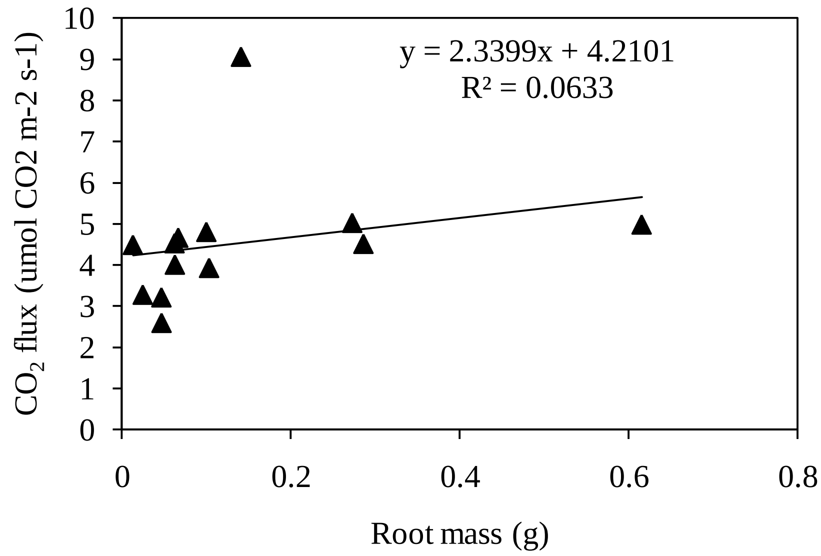


Figure B.2 Root mass regression plot for the Te Kowhai soil (i.e. all paddocks plotted). Hetrotrophic respiration which is determined from Y axis intercept (i.e. zero root mass) contributed 94% to average respiration for the soil of 4.5 µmol CO_2 m^{-2} s^{-1}.

Discussion

Root mass regression works by plotting the root mass of a sample against its average CO_2-C flux. A positive relationship between root mass and soil respiration is expected as the greater the mass of live roots within soil the greater the contribution of autotrophic respiration to total soil respiration. By collecting numerous samples with varying root mass, a regression plot can be produced that will allow the CO_2-C flux for soil with zero root mass to be determined by reading off the Y axis intercept of the regression line. To improve the performance of this method the following recommendations can be made;

- Collect a minimum of 20 samples per hectare
- Separate live roots and discard dead roots
- Soak soil and apply calgon to disperse clay from roots as any debris retained on the roots will be included in the root mass calculation

APPENDIX C

Gap filling and flux partitioning

Gaps produced following filtering of 30 minute net ecosystem CO_2 exchange (NEE) and latent heat flux density (λE) were filled using the online gap filling model presented by Reichstein *et al.* (2005). This software was also applied to partition NEE into gross primary productivity (GPP) and total ecosystem respiration (TER). Moffat *et al.* (2007) reviewed 15 methods for gap filling 30 minute NEE and concluded that the approach of Reichstein *et al.* (2005) displayed good overall performance, because of this the method has been adopted as one of two standardized gap filling techniques by the Carboeurope IP project and FLUXNET (Moffat *et al.* 2007).

The online gap filling model can be accessed at;

http://gaia.agraria.unitus.it/database/carboeuropeip/

By clicking under "data" and then "online gap filling model". The model is described fully on the previously mentioned website and in the paper by Reichstein *et al* (2005).

Gap filling

The online gap filling model of Reichstein *et al* (2005) is similar to the methods of Falge *et al.* (2001), the main difference being that the online gap filling model considers both the covariation of fluxes with meteorological variables and the temporal auto-correlation of the fluxes (Reichstein *et al.* 2005). When only NEE data were missing but all other meteorological data were available, then missing NEE data would be replaced by the average NEE value under similar meteorological conditions (i.e. global radiation (Rg) within 50 W m^{-2}, air temperature (T_{air}) within 2.5 °C, and vapour pressure deficit (VPD) is within 5.0 hPa) within a time window of ± 7 days. If similar meteorological conditions are not available in the ± 7 day window, the averaging window is increased to ±14 days. If T_{air}, VPD and NEE are missing the averaging window is maintained at ± 14 days and similar meteorological conditions are identified using Rg. If Rg is also missing, then NEE values were filled by applying the mean diurnal variation technique which starts with a window size of ± 0.5 days if the gap is not filled then the window size increases until all gaps in NEE are filled.

Flux partitioning

The online flux partitioning model of Reichstein *et al* (2005) is based around the Lloyd and Taylor (1994) regression model. Initially the components of the dataset were divided into day-time and night-time values based on a night-time global radiation (Rg) threshold of <20 W m^{-2}, and a day-time threshold of >20 W m^{-2}. Night-time data was then divided into consecutive 10 day windows and the Lloyd and Taylor (1994) regression model was fitted to NEE (GPP = 0 at night, therefore, night-time NEE is equivalent to TER) and air temperature data for each of the 10 day windows. Model parameters from night-time data were then applied in combination with daytime air temperature to predict daytime TER. Gross primary production (GPP) was then calculated by subtracting NEE from TER.

APPENDIX D

Energy balance closure

Figure D.1 and D.2 were generated using filtered half hourly values of λ*E* and *H* measured at the site for 2008 (15/12/07 – 14/12/08) and 2009 (15/12/08 – 23/11/09). Energy balance closure was 12.2% and 9.3% for 2008 and 2009 respectively.

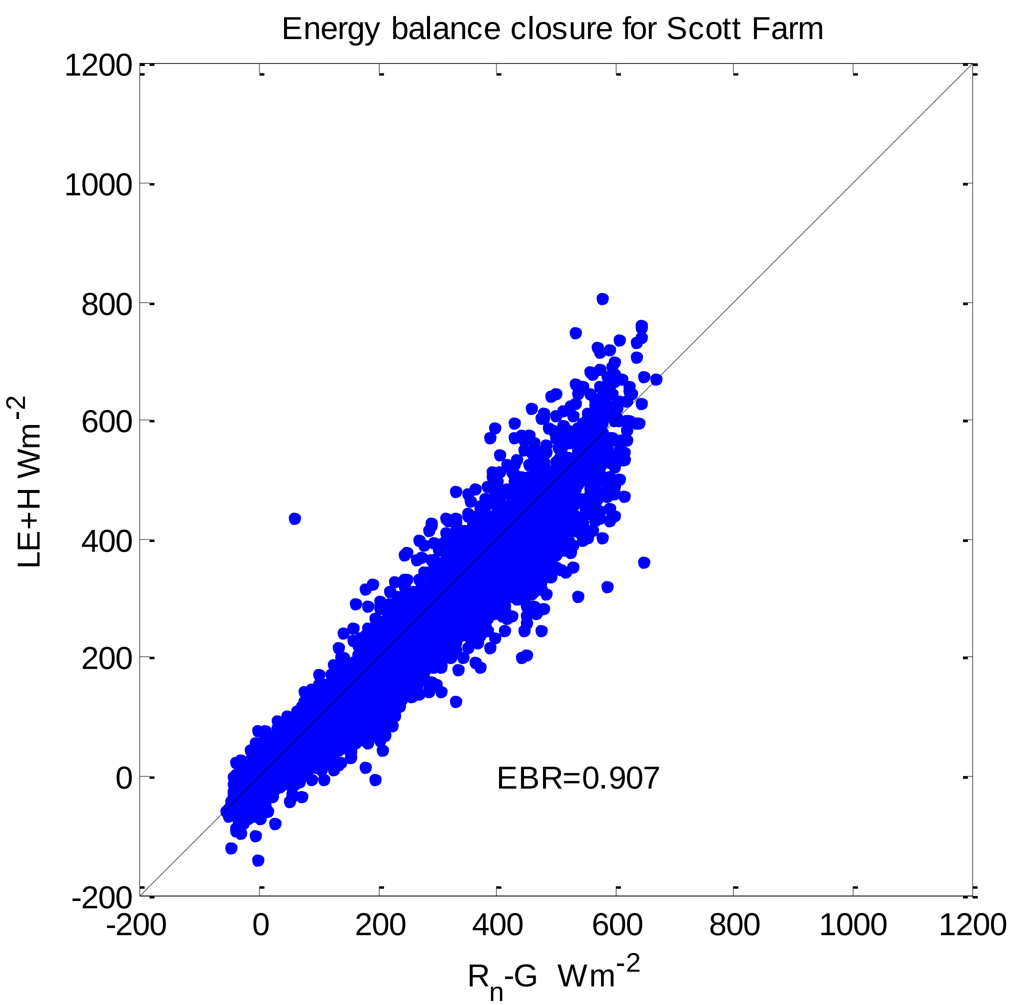


Figure D.1 30 minute energy balance closure at Scott Farm for the 2009 year (15/12/08 – 23/11/09)

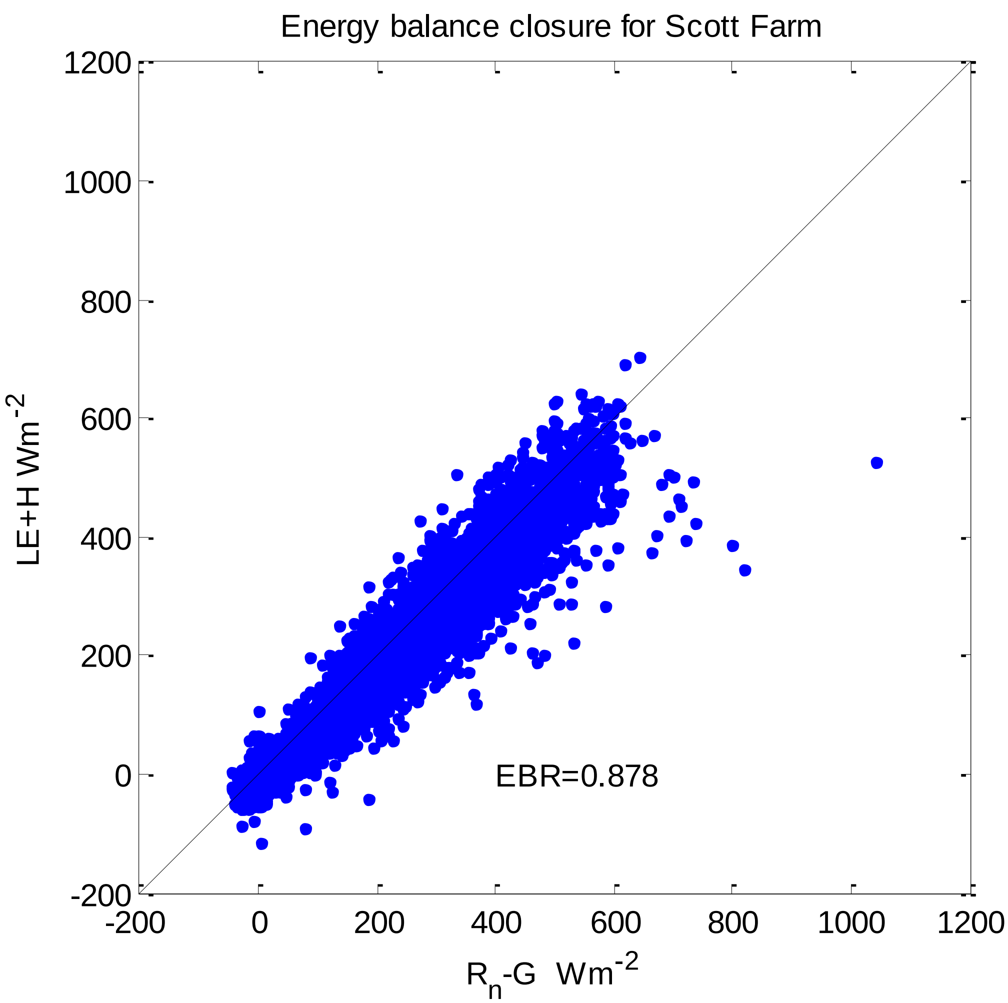


Figure 2.D 30 minute energy balance closure at Scott Farm for the 2008 year (15/12/07 – 14/12/08)

APPENDIX E

Digital appendices

The attached CD-ROM contains relevant information referred to throughout the thesis. The disc contains;

- Photographs of the EC and cultivation sites and equipment used for root washing.
- Sigma plot figures.
- Matlab scripts.
- Raw data from cultivation study.