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GHG emissions from managed peat soils

Final Report prepared for Ministry for Primary Industries

June 2014

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1. Executive Summary

This final report includes the contracted outputs (manuscripts) from two separate, yet linked, field studies which have quantified greenhouse gas (GHG) emissions from managed peat soils including nitrous oxide (N_2O) and carbon dioxide (CO_2). AgResearch measured background N_2O emissions (E_{N2O}) at a grassland site (dairy farm) with peat soil in the Waikato region. The measurements were made repeatedly at ten locations within the site throughout a year at fortnightly intervals. By fencing, the measurement site was not grazed and no nitrogen fertiliser was applied during the study. For a measurement set in winter, the mean E_{N2O} reached a maximum rate equivalent to 30.8 kg N ha^{-1} y^{-1} when groundwater came within 0.4 m of the surface, while in summer, E_{N2O} was generally equivalent to 0.2 – 0.4 kg N ha^{-1} y^{-1} and depth to groundwater 1.0 – 1.7 m. There was an asymptotic exponential relationship between E_{N2O} and the depth to groundwater. During 100 winter days when the cumulative rainfall was 342 mm, E_{N2O} was also measured fortnightly in a covered area. Under the cover, the soil's nitrogen content averaged 5-fold larger and the E_{N2O} 3-fold larger than in the uncovered area, reflecting a rainfall effect on soil nitrogen availability and E_{N2O}. The time series of E_{N2O} data in the uncovered area was variable and the frequency distribution skewed. Consequently, we needed to 'normalise' these data for statistical analysis using $\log_e$ transformed values. In addition, to appropriately take account of the correlation between the fortnightly, repeated measurements, we needed to use an order 2 autoregressive model which was selected based on the Akaike Information Criterion. The back-transformed grand mean E_{N2O} value was equivalent to 1.6 kg N ha^{-1} y^{-1} and the estimated lower and upper 95% confidence limits were 0.4 and 7.0 kg N ha^{-1} y^{-1}, respectively. Statistically, a grand mean E_{N2O} from this study was significantly less than 8 kg N ha^{-1} y^{-1}, recommended by the Intergovernmental Panel on Climate Change for Tier 1 inventories for countries with managed peat soils and a temperate climate. Thus, for New Zealand's inventory of nitrous oxide emissions from agricultural soils, we recommend the emissions factor for managed peat soils should be reduced from 8 to 1.6 kg N ha^{-1} y^{-1}. The uncertainty of this recommended emission factor should be estimated by the 95% confidence limits determined from this study.

The subcontracted University of Waikato science team focused on the causes and magnitudes of spatial and temporal variability of CO_2 fluxes from dairy farms on deep peat, including the impacts of grazing and variations of drainage management. To identify and account for the large changes in pasture biomass that accompany intensive grazing, a "phytomass index" was calculated from eddy covariance measurements made over four dairy farms on peat, and the improvement to ecosystem models of CO_2 fluxes tested.

A re-analysis of previously published eddy covariance measurements at one of the farms over 12 months showed the pasture system was a net source of CO_2 equivalent to 2060 kg C ha^{-1} y^{-1}, substantially more than was previously reported, but now in agreement with mass loss calculations for Waikato peats and the published literature on northern hemisphere agricultural peatlands.

After comparing measurements made at pairs of farms, there appeared to be no significant differences in night-time CO_2 emissions between farms, despite some large differences in depth to water table during the dry late summer and autumn seasons. However, substantial differences in daytime CO_2 fluxes under non-limiting light conditions (standardised between sites by each farm's phytomass index) were evident during these dry periods, suggesting that the limiting effect of low peat soil moisture content was driving large spatial differences in pasture growth more than soil respiration rates. An understanding of the impact of different water table depths on photosynthesis and pasture growth is critical if estimates of annual losses of CO_2 from the full range of farmed peats are to be made. However, if the water table effect is confirmed, an important management implication was that maintaining a higher water table would likely decrease CO_2 losses and also increase pasture growth. Identifying and managing soils to maintain an optimum water table depth could reduce CO_2 losses from dairy farms and enhance pasture production during dry late summer and autumn periods.

2. Background

For New Zealand's inventory of nitrous oxide (N_2O) emissions from agricultural soils, a Tier 1 approach estimates the flux (F_{N2O}) from 285,629 hectares of lowland, managed peat soils beneath pasture, mostly located in the Waikato region. The emission factor for this land has been set to the default value of F_{N2O} equal to 8 kg N ha^{-1} y^{-1}. This F_{N2O} is recommended for countries with managed peat soils and a temperate climate according to the Intergovernmental Panel on Climate Change (IPCC) in the 2006 Guidelines for national greenhouse gas inventories (Volume 4, Chapter 2, Equation 11.1). To determine if this recommendation is appropriate for New Zealand, F_{N2O} measurements were needed from managed, peat soils beneath pasture. AgResearch made these measurements at a representative site and prepared a manuscript on the results. The manuscript was submitted for publication to the international, scientific, peer-reviewed journal Agriculture, Ecosystems and Environment. The manuscript is included with this final report in Appendix 1.

Peat soils also exchange the greenhouse gas carbon dioxide (CO_2) with the atmosphere, but uncertainty also makes it difficult to determine this exchange rate on an annual basis, requiring more measurements and modelling. Using the micrometeorological eddy covariance method, paddock-scale measurements can be made of CO_2 exchange between the atmosphere and managed peat soils. There has been one study which compiled an annual CO_2 and overall farm-scale carbon (C) budget for a dairy farm on peat soil in NZ, at Rukuhia near Hamilton (Nieveen et al., 2005. Global Change Biology 11: 607–618). The annual CO_2 flux was close to zero, while C exports from the farm by way of enteric methane emissions and milk products implied a net loss of soil C of around 1060 kg C ha^{-1} y^{-1}. In contrast, at a nearby location, Schipper & McLeod (2002. Soil Use and Management 18: 91–93) measured the C content of soil/peat cores in both farmland and an adjacent peat bog and found net soil C loss of 3700 kg C ha^{-1} y^{-1} averaged over 40 years since the land was converted.

The University of Waikato science team were subcontracted by AgResearch to re-analyse their eddy covariance and environmental measurements made at four dairy farms with peat soils and drainage ditches of different depths, focusing on the causes and likely magnitudes of spatial and temporal variability of CO_2 exchange. They were also subcontracted to prepare a manuscript on the results. The manuscript was submitted for publication to the international, scientific, peer-reviewed journal Agriculture, Ecosystems and Environment. The manuscript is included with this final report in Appendix 2.

3. Appendix 1: Background nitrous oxide emissions from grassland with peat soil in New Zealand - manuscript submitted to the international journal Agriculture, Ecosystems and Environment by Kelliher et al.

3.1 Introduction

In accordance with commitment to the United Nations Framework Convention on Climate Change (UNFCCC), participating countries produce annual greenhouse gas emissions inventories including nitrous oxide (N_2O) emissions (E_{N2O}) from agricultural land. Following the Intergovernmental Panel on Climate Change (IPCC) guidelines, for managed organic (i.e., peat) soils, E_{N2O} can be attributed to the effects of (i) nitrogen (N) application and (ii) cultivation and drainage (de Klein et al. 2006). Fertiliser N can be applied to soils and in pastoral agriculture, N is returned to soils by grazing animals. Accordingly, the greater the fertiliser and/or excreta N application and/or deposition, the greater will be E_{N2O}. Separately, cultivation and drainage of managed peat soils are considered to induce a background E_{N2O} (e.g., Maljanen et al. 2007). The IPCC recommends a background E_{N2O} (hereafter, E_{N2O}) of 8 kg N ha^{-1} y^{-1} in so-called Tier 1 inventories for managed peat soils in countries with a temperate climate (de Klein et al. 2006). We interpreted the recommended value to be a mean based on E_{N2O} measurements from field studies. In addition to the mean, de Klein et al. (2006) also reported a minimum of 2 kg N ha^{-1} y^{-1} and maximum of 24 kg N ha^{-1} y^{-1}. On this basis, a difference between the minimum and mean was 6 kg N ha^{-1} y^{-1}, while a difference between the maximum and mean was 16 kg N ha^{-1} y^{-1}. Thus, we interpreted the E_{N2O} data available to de Klein et al. (2006) were skewed. These interpretations of a wide range and skewed distribution were supported by the results of continuous, long-term E_{N2O} measurements from peat soils beneath managed grassland (Scanlon and Kiely 2003; Kroon et al. 2010).

The purpose of this study was to determine the suitability of 8 kg N ha^{-1} y^{-1} as an inventory E_{N2O} estimate for managed grassland with peat soil in New Zealand. We identified a representative, suitable site and measured E_{N2O} at two-weekly intervals throughout a year. To better understand the time course of E_{N2O}, we concurrently measured environmental variables. Moreover, during winter when rainfall is likely to be greatest and groundwater closest to the surface, we made additional E_{N2O}, soil N content and environmental measurements in adjacent plots, one uncovered and the other covered to exclude rainfall. Here, we report the results, statistically analyse them and make comparisons with the available, published data.

3.2 Materials and Methods

The measurements were made about 15 km northeast of the Ruakura Research Centre near the city of Hamilton, New Zealand (37.7 °S, 175.4 °E). The measurement site was located on a commercially-managed dairy farm of 165 hectares divided by fencing into paddocks for rotational grazing by the cattle. The vegetation was permanent pasture comprised of predominantly perennial ryegrass (*Lolium perenne* L.) and white clover (*Trifolium repens* L.). The measurement site was last grazed on 29 September 2012, and afterwards, a 25 m by 15 m area was delineated by fencing to exclude the cattle. Within the fenced area, the grass was mown regularly to a height of 2.5 cm and no fertiliser was applied to the soil. About 100 m east of this area, there was a 2-m-deep drain that had recently been cleared. A soil sample consisting of (10) cores collected to a depth of 7.5 cm across the measurement area had an organic carbon content of 31.7 g C kg^{-1}, total N content of 1.7 g N kg^{-1}, cation exchange capacity of 67 cmol kg^{-1} and pH of 6.6 (water). By digging pits in the measurement area and examining the nearby drain, we concluded that the peat depth exceeded 1 m. The soil's name was Kaipaki peat loam which is a Mellow Mesic organic soil according to the New Zealand soil classification system (Hewitt 2010). Before the farm's land was drained for dairying approximately 60 years ago, the former bog vegetation included wire rush (*Empodisma minus*) and the endemic cane rush (*Sporadanthus ferrugineus*), their roots forming the bulk of the peat (Clarkson and Clarkson 2006).

Within the fenced area, ten locations were established for measuring E_{N2O}. At each location, we inserted a chamber base (each 12 cm long by 23.6 cm diameter) to a depth of 10 cm. The chamber bases were located along a linear transect at 1 m intervals. Beginning 11 January 2013, and thereafter at fortnightly intervals for the rest of the year, each time around the middle of the day, a static chamber was placed onto each base with a gas-tight seal, so air samples could be collected during an enclosure period (described below). To re-iterate, the cattle were excluded for 3 months prior to the first E_{N2O} measurement and it had been four months since N fertiliser (25 kg N ha^{-1}) had been applied.

Before winter (June – August), when rainfall is likely to be greatest, a 10 m long by 2 m wide plastic roof was erected within the fenced area. Under the roof, ten more chamber bases were inserted along another linear transect at 1 m intervals. Beginning 23 May, and thereafter on 7 and 21 June, 19 July and 5, 20 and 30 August, a static chamber was placed onto each base under the roof for air sampling. At the same time on these dates, a static chamber was also placed onto each base in the uncovered area for air sampling. On these dates, for water, nitrate (NO_3^-–N) and ammonium (NH_4^+–N) content measurements, soil samples were collected at multiple locations using a steel tube inserted to a depth of 7.5 cm under the roof and in the uncovered area.

To measure the NO_3^-–N and NH_4^+–N contents, the soil was extracted using 2M KCl (1:10) by shaking for 1 hour on an end-over-end shaker and filtering (Whatman 42, Mulvaney1996). The filtered soil extracts were analysed for NO_3-N and NH_4-N using a Skalar SAN^{++} segmented flow analyser (Skalar Analytical B.V., Breda, Netherlands). The NO_3^-–N method involves cadmium reduction to nitrite followed by diazotisation with sulphanilamide and coupling with N-(1-naphthyl) ethylenediamine dihydrochloride to form an azo die measured colourimetrically at 540 nm. The NH_4^+–N method is based on the modified Berthalot reaction. The NH_4^+–N is chlorinated to monochloramine which reacts with salicylate and is then oxidised to form a blue/green coloured complex which is measured colourimetrically at 660 nm. The auto-analyser had a detection limit for NO_3^-–N and NH_4^+-N of 0.4 mg L^{-1}. The NO_2^--N content was assumed to be close to the detection limit in all samples, so the results reported as NO_3^--N contents actually comprised a sum of the NO_2^--N and NO_3-N contents. To measure the gravimetric water content, the soil sample was weighed, dried at 105°C for 24 h and weighed again (McLay et al. 1992). For each sample, the bulk density was calculated from the sample's dry weight, the sampled depth and (inside) diameter of the sampling tube. The porosity was calculated from the bulk density using a particle density of 1560 kg m^{-3} following McLay et al. (1992).

Air samples were collected using a syringe via a tube fitted with a tap attached to the top of the static chamber. The sample volume of ~10 mL was transferred through a 3-mm-thick, butyl rubber septa into an evacuated 6 mL vial (Exetainer®, Labco Limited, Lampeter, UK), over-pressurised to minimise the incursion of ambient air during storage until analysis. From each chamber, three air samples were taken at 0.5 h intervals, after which the chamber was removed. By calculation, $E_{N2O} = (\Delta c/\Delta t)*(M/V_m)*(V/A)$ where Δc is the change of N_2O concentration in the chamber headspace during an enclosure period (µL L^{-1}), Δt the enclosure period, M the molar mass of N in N_2O (g/mol), V_m the molar volume of gas at the sampling temperature and atmospheric pressure (L/mol), V the headspace volume (m^3) and A the area covered (m^2).

Within one week of collection, the N_2O concentration in an air sample was measured by a gas chromatography (GC) system similar to that described by Mosier and Mack (1980) with some distinguishing features described here briefly. The vial containing a sample replaced the fixed-volume sample loop normally employed for gas analysis, and the entire 6 mL sample volume was injected into the GC. Prior to GC analysis, the pressure within a vial was equilibrated to the ambient atmospheric level. Each vial, containing the de-pressurised air sample, was then placed in the rack of an auto-sampler (Gilson 222XL, Middleton, WI, USA) interfaced to a GC (model 8610, SRI Instruments, San Francisco, CA, USA) using N_2 as the carrier gas. A purpose-built double concentric injection needle (PDZ-Europa, Crewe, UK) pierced each vial's septa and rapidly flushed the air into GC columns packed with Haysep Q (OD 3 mm, pre-column of length 1.0 m and analytical column, 6.0 meter) set at an operation temperature of 35°C. The N_2O concentration was quantified by a ^{63}Ni electron capture detector operated at 320°C. The GC system was calibrated using an appropriate selection of standards with N_2O concentrations of 0 (N_2 gas), 0.32 (air), 0.50, 1.00, 2.00, 5.0, 10.0, 20.0 and 50.0 μL L^{-1}. The GC system precision was assessed regularly by measuring the N_2O concentration of 20 air samples (0.32 μL L^{-1}), and typically, the standard deviation was 1% of the mean, 2% for 95% confidence.

Throughout the year, rainfall was measured and recorded daily at Ruakura. An earlier 40 year record (beginning 1 January 1973) was analysed statistically. For each month, total rainfall data were sorted from smallest to largest values to determine percentile and mean values. In the fenced area with the chamber bases, to measure the depth to groundwater, we pushed a 2-m-long perforated pipe, covered with fine-mesh gauze, into the soil. Each time air samples were collected for measuring E_{N2O}, we lowered an electronic device into the pipe and upon reaching the groundwater, an electronic circuit closed, an alarm sounded and the depth was measured and recorded.

3.3 Results

For the year 2013, annual rainfall was 980 mm on 143 days. Based on the 40-year record, this annual rainfall was virtually identical to the 10^{th} percentile, and the mean annual rainfall was 1152 mm. During the first three months of 2013, the summer season, only 50 mm of rain fell on 8 days (Table 1). For context, the estimated evaporation from well-watered pasture for this period was 330 mm (Scotter and Heng 2003). Each of these three monthly rainfalls was significantly less ($P < 0.05$) than the corresponding 40-year means which averaged 77 mm $month^{-1}$. During the remaining nine months of 2013, rain fell every other day, on average, and rainfall exceeded evaporation by 360 mm.

For the 27 fortnightly sets of measurements, the mean E_{N2O} varied by more than 100-fold to a maximum equivalent to 30.8 kg N ha^{-1} y^{-1} on 7 June (Table 1). We estimated the minimum E_{N2O} which could be reliably measured was equivalent to 0.3 kg N ha^{-1} y^{-1}. For this calculation, we used the E_{N2O} equation with Δc set to the GC precision for 95% confidence. During the first three months of 2013, in conjunction with the low rainfall and dry soil (data not shown), the calculated mean E_{N2O} was equivalent to a value significantly greater than 0.3 kg N ha^{-1} y^{-1} only once, on 20 March. This result was evidently induced by 23 mm of rain on 18 and 19 March after 40 days without rain beforehand. After March, each fortnightly mean E_{N2O} was equivalent to a value significantly greater than 0.3 kg N ha^{-1} y^{-1} until 22 November, the following summer.

In addition to temporal variability of the E_{N2O} measurements, each fortnightly set was spatially variable. We calculated correlation coefficients from the 27 sets of measurements to determine if any of this variability could be attributed to the proximity of one chamber to another. These results suggested no spatial pattern of E_{N2O} (data not shown).

Based on the 27 fortnightly sets of E_{N2O} measurements, our first calculation of an overall (grand) mean gave a value equivalent to 4.8 kg N ha^{-1} y^{-1}. This conventional calculation assumed each set was independent. However, on the basis of each set of measurements being repeated at the same ten locations on different days (i.e., the measurements are a time series), correlation of the sets seems likely and a conventional calculation would not be appropriate. We also calculated a grand median equivalent to 1.6 kg N ha^{-1} y^{-1} which was substantially different to 4.8 kg N ha^{-1} y^{-1}. This suggested a skewed distribution for the fortnightly sets of E_{N2O} measurements. Therefore, further statistical analysis used $\log_e$ (base 2.718) transformed values in order to 'normalise' these data (i.e., to make their distribution closer to a normal, bell-shaped distribution). Then, the time series of E_{N2O} measurements was analysed using an order 2 autoregressive model to determine an appropriate grand mean E_{N2O} by taking account of the correlation between the sets. Selection of this autoregressive model was based on the Akaike Information Criterion. This resulted in a $\log_e$ grand mean and accompanying $\log_e$ standard error of 0.50 and 0.64, respectively. Thus, a back-transformed, appropriately-calculated grand mean was 1.6 kg N ha^{-1} y^{-1} with back-transformed 95% confidence limits of 0.4 and 7.0 kg N ha^{-1} y^{-1}. This grand mean is identical to the grand median and substantially less than the conventional grand mean of 4.8 kg N ha^{-1} y^{-1}, which was calculated assuming each set was independent. We next tested a null hypothesis that the appropriately-calculated grand mean of 1.6 kg N ha^{-1} y^{-1} was equal to 8 kg N ha^{-1} y^{-1}, the IPCC-recommended value for managed peat soils for Tier 1 inventories of countries with a temperate climate (de Klein et al. 2006). For this purpose, a one-sample T test (n = 10) was performed on $\log_e$ transformed values as each set of E_{N2O} measurements was made at ten locations. The calculated *P* value was 0.035, indicating sufficient evidence to reject the hypothesis with 95% confidence level ($P < 0.05$).

From 23 May – 30 August (100 days in winter) when E_{N2O} was measured in the uncovered and covered areas, the cumulative rainfall was 309 mm and the mean depth to groundwater 0.50 m. During this period, the uppermost 7.5 cm depth of soil in the uncovered area was substantially wetter than under the cover (Table 2). Moreover, under the cover, the soil's mean NO^-_3–N content was nearly 6-fold greater and the NH^+_4-N content 4-fold greater than in the uncovered area. The corresponding mean E_{N2O} was equivalent to 33 kg N ha^{-1} y^{-1} under the cover and 11 kg N ha^{-1} y^{-1} in the uncovered area.

3.4 Discussion

During June – August, winter, E_{N2O} was by far the greatest and the depth to groundwater (depth, m) least. By regression analysis using $\log_e$ transformed E_{N2O} values against groundwater depth with an asymptotic exponential function, we were able to meet the assumptions of constant variance and normality of the residuals. The resultant relationship was $\log_e E_{N2O} = a - be^{(-c*depth)}$ where parameters a, b and c (± standard error) are -0.87 ± 0.37, -16.60 ± 9.49 and 3.65 ± 1.36, respectively (Figure 1). The standard error of an E_{N2O} estimate was 1.0 kg N ha^{-1} y^{-1}, equivalent to 20% of the mean E_{N2O}. There was an inflection in the relationship when the depth to groundwater was about 0.5 m. At another dairy farm located ~30 km southwest of our study, for depth to groundwater < 0.5 m, water content in the uppermost 0.3 m depth of the peat soil was constant and maximal, while for depth > 0.5 m, the water content declined sharply and linearly (Nieveen et al. 2005, see their Figure 4). Such thresholds reflect capillary rise (water flow) in soils, controlled by a relationship between hydraulic conductivity and pore size and the pore size distribution (Baird 1997).

The soil's N content and turnover rate are additional variables which can affect E_{N2O}. From 23 May – 30 August (100 days in winter) when E_{N2O} was measured in the uncovered and covered areas, the mean soil temperature was 12 °C. For this temperature, the estimated N turnover was 8% according to Clough et al. (1998), expressing net N mineralisation as a percentage of total soil N (NO^-_3 + NH^+_4). Under the cover, as indicated, the mean NO^-_3–N content was nearly 6-fold greater and the NH^+_4-N content 4-fold greater than in the uncovered area. On these bases, soil N availability and E_{N2O} were substantially greater under the cover.

In Germany, four studies of E_{N2O} from grassland sites with peat soil indicated highly variable results (Table 3). The largest mean, equivalent to 29.0 kg N ha^{-1} y^{-1}, came from a drained meadow site in Donaumoos, a fen located ~80 km north of Munich, Germany (Flessa et al. 1998). A corresponding, reported median of these 41 sets of measurements was equivalent to 2.0 kg N ha^{-1} y^{-1}. Moreover, another mean was equivalent to 5.3 kg N ha^{-1} y^{-1} from a different drained meadow site in Donaumoos during a different year (Wild et al. 2001). The mean was also equivalent to 5.3 kg N ha^{-1} y^{-1} from drained grassland at Rhil-Havelluch, another fen located near Brandenburg, Germany, ~400 km northeast of Donaumoos (Augustin et al. 1998). By contrast, the mean was equivalent to only 0.9 kg N ha^{-1} y^{-1} from drained grassland at The Grosses Moor, a peat bog located near Gifhorn, Germany, about 550 km north of Donaumoos (Leiber-Sauheitl et al. 2014).

The mean E_{N2O} from grassland sites with peat soil near Kannus, Finland and Bodo, Norway was equivalent to 6.3 and 8.2 kg N ha^{-1} y^{-1}, respectively (Maljanen et al. 2007; Kløve et al. 2010). These sites had a boreal climate (mean air temperature about 3 - 4°C) which evidently did not induce an exceptional E_{N2O} compared to the mean E_{N2O} from grassland sites with peat soil and a temperate climate (Table 3). For our New Zealand site, the mean air temperature was about 13°C.

In other studies, the mean E_{N2O} from ten grassland sites with peat soils ranged widely from equivalence to 0.9 kg N ha^{-1} y^{-1} (Leiber-Sauheitl et al. 2014) up to 29.0 kg N ha^{-1} y^{-1} (Flessa et al. 1998)(Table 3). These minimum and maximum values are broadly similar to those determined by the IPCC for countries with a temperate climate and managed peat soils (de Klein et al. 2006). However, the data from six of the ten grassland sites were not available to de Klein et al. (2006), having been published afterwards. From the (untransformed) means in Table 3, excluding the one from this study, we used conventional calculations to compute a grand mean equivalent to 7.4 kg N ha^{-1} y^{-1} and a median of 5.3 kg N ha^{-1} y^{-1}. Both of these central tendency estimates are substantially greater than a grand mean equivalent to 1.6 kg N ha^{-1} y^{-1} from this study.

3.5 Conclusions

In summer, E_{N2O} was generally equivalent to the detection limit of 0.3 kg N ha^{-1} y^{-1} when rainfall was sparse and depth to groundwater 1.0 – 1.7 m. In winter, E_{N2O} increased to the equivalent of 30.8 kg N ha^{-1} y^{-1} when rainfall was greatest and groundwater came within 0.4 m of the surface. There was an asymptotic exponential relationship between E_{N2O} and the depth to groundwater. During winter when temperature averaged 12°C and rainfall exceeded 100 mm $month^{-1}$, compared to soil in a covered area, the uncovered soil's nitrogen content was 5-fold smaller and the E_{N2O} 3-fold smaller, reflecting temperature and rainfall effects on soil nitrogen availability and E_{N2O}. The time series of E_{N2O} data in the uncovered area was variable and the frequency distribution skewed. Consequently, we needed to 'normalise' these data for statistical analysis using $\log_e$ transformed values. In addition, to appropriately take account of the correlation between the fortnightly, repeated measurements, we needed to use an order 2 autoregressive model which was selected based on the Akaike Information Criterion. The back-transformed grand mean E_{N2O} was equivalent to 1.6 kg N ha^{-1} y^{-1} and the estimated lower and upper 95% confidence limits were 0.4 and 7.0 kg N ha^{-1} y^{-1}, respectively. Statistically, a grand mean E_{N2O} from this study was significantly less than 8 kg N ha^{-1} y^{-1}, recommended by the Intergovernmental Panel on Climate Change for Tier 1 inventories of countries with a temperate climate.

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Table 1 Background nitrous oxide (N_2O) emissions (E_{N2O}, expressed equivalent to an annual basis, kg N ha^{-1} y^{-1}) measured repeatedly at ten locations in the uncovered area throughout a year at fortnightly intervals (mean ± standard error, SE). Also shown is the depth to groundwater (m) and cumulative rainfall (mm) from 1 January 2013.

Date in 2013	E_{N2O}	Groundwater depth	Cumulative rainfall
11 January	0.41 ± 0.11	1.03	4
25 January	0.22 ± 0.06	1.18	7
8 February	0.47 ± 0.09	1.32	27
25 February	0.12 ± 0.03	1.54	27
11 March	0.15 ± 0.04	1.64	27
20 March	1.28 ± 0.22	1.68	50
28 March	0.25 ± 0.06	1.70	50
5 April	1.63 ± 0.24	1.64	59
19 April	0.78 ± 0.11	1.57	79
26 April	1.75 ± 0.14	1.35	166
10 May	1.62 ± 0.36	1.30	217
23 May	6.69 ± 1.47	0.41	316
7 June	30.85 ± 4.04	0.51	387
21 June	24.79 ± 8.37	0.36	514
5 July	18.39 ± 1.59	0.45	519
19 July	5.09 ± 0.43	0.55	540
5 August	1.03 ± 0.14	0.64	547
20 August	3.10 ± 0.98	0.52	570
30 August	5.48 ± 0.96	0.51	610
16 September	4.55 ± 1.06	0.50	672
1 October	11.41 ± 2.52	0.52	747
16 October	5.64 ± 1.89	0.60	796
25 October	0.73 ± 0.16	0.68	798
8 November	0.76 ± 0.22	0.76	849
22 November	0.13 ± 0.04	0.93	868
9 December	2.96 ± 0.62	0.88	962
23 December	0.24 ± 0.08	1.09	962

Table 2 Mean values of background nitrous oxide (N_2O-N, nitrogen) emissions (E_{N2O}, ± SEM, expressed equivalent to an annual basis, kg N ha^{-1} y^{-1}) in the uncovered and covered areas from seven sets of measurements between 23 May and 30 August 2013. Also shown for the uppermost 7.5 cm depth of soil are corresponding values of volumetric water content (m^3 m^{-3}) and the nitrate (NO^-_3, mg N kg^{-1}) and ammonium (NH^+_4, mg N kg^{-1}) contents. For context, the soil's porosity was 0.78 m^3 m^{-3}.

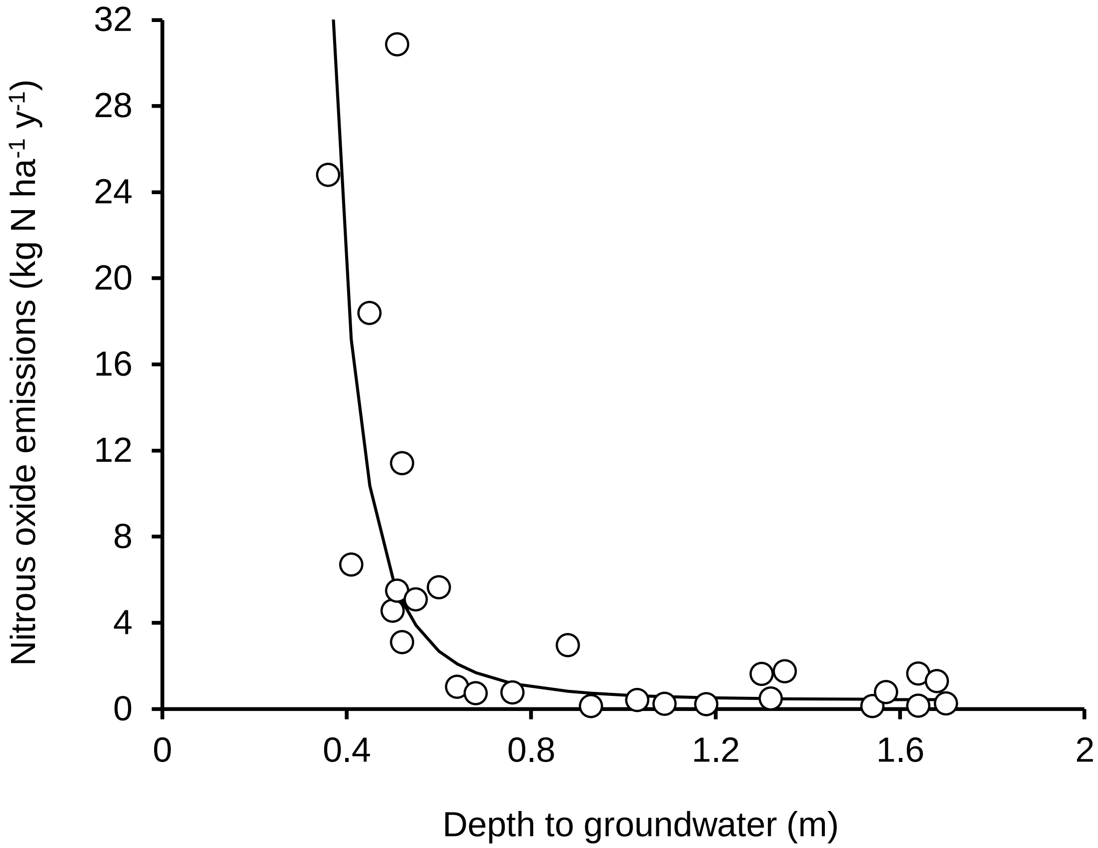
Treatment	Uncovered area	Covered area
E_{N2O}	11.0 ± 2.2	33.0 ± 8.0
Water content	0.42 ± 0.01	0.18 ± 0.04
NO^-_3 content	13.4 ± 3.0	74.4 ± 8.8
NH^+_4 content	9.0 ± 2.9	40.2 ± 9.2

Table 3 Background nitrous oxide (N_2O-N, nitrogen) emissions (E_{N2O}, mean ± standard deviation, expressed equivalent to an annual basis, kg N ha^{-1} y^{-1}) at grassland sites with peat soils (sum of the number of sites and years of measurement in brackets). Also shown are citations for the data sources. For this study, using $\log_e$ transformed values which 'normalised' the data and an order 2 autoregressive model, we determined a (grand) mean E_{N2O} based on the Akaike Information Criterion as explained in the text.

Country	E_{N2O}	Source
Netherlands	5.2 ± 5.2[a] (4)	Velthof et al. 1996
Netherlands	3.1 ± 2.8[a] (8)	Van Beek et al. 2011
Germany	5.3 (1)	Augustin et al. 1998
Germany	29.0[b] (1)	Flessa et al. 1998
Germany	5.3[b] (1)	Wild et al. 2001
Germany	0.9 ± 0.2 (6)	Leiber-Sauheitl et al. 2014
Finland	8.2 ± 8.8 (10)	Maljanen et al. 2007
Norway	6.3 (1)	Kløve et al. 2010
Sweden	7.1 ± 3.8 (4[c])	Berglund and Berglund 2011
Denmark	3.2 ± 1.7[d] (9)	Petersen et al. 2012
New Zealand	1.6 (1)	This study

[a]These measurements were made at the same sites in different years
[b]These measurements were made in Donaumoos Mire at different sites in different years and the data reported by Wild et al. (2001) were extrapolated from 9 to 12 months
[c]Soil monoliths from two sites, each subjected to two water-level treatments
[d]From their Table 8, the 95% confidence limits were $\log_e$ transformed, an average difference from the mean calculated, divided by two and this result was back-transformed to estimate a standard deviation

Figure 1 Relationship between the mean background nitrous oxide (N_2O-N, nitrogen) emissions, measured at fortnightly intervals throughout a year but expressed equivalent to an annual basis, and the corresponding depth to groundwater in the uncovered area. The curve is an asymptotic exponential function fitted by regression to $\log_e$ transformed values of these data as described in the text.



4. Appendix 2: Variations in CO_2 exchange for dairy farms with year-round rotational grazing on peat soils – a manuscript to the international journal Agriculture, Ecosystems and Environment by Campbell et al.

4.1 Introduction

Globally peatlands represent 3% of land surface area yet store carbon (C) equivalent to 30% of the global soil C stock (Blodau, 2002). The C stored in peatlands has accumulated over thousands of years because of net CO_2-C uptake via photosynthesis exceeding C losses via decomposition processes (Frolking et al., 2011), and their overall greenhouse gas (GHG) balance has had a net cooling effect on global climate throughout the Holocene, despite being significant sources of methane (Frolking and Roulet, 2007). In contrast, peatlands developed for agriculture are generally CO_2 sources due to lowering of the water table and improved soil fertility altering the balance between gross primary productivity (GPP) and ecosystem respiration (ER). Lowered peatland water tables leads to altered peat decomposition rates, both directly via a deeper layer of peat exposed to oxygen, and indirectly because of changes to microbial populations (Mäkiranta et al., 2009). Because human activities have led to net CO_2 losses within drained peatlands and, usually, increased N_2O emissions, they must be accounted for in national greenhouse gas inventories (Couwenberg, 2011).

While conversion of natural ecosystems for agriculture alone has had major impacts on the net ecosystem exchange of CO_2 (NEE=ER–GPP), ongoing management of agricultural systems involve an increase in disturbance regimes that can lead to additional alteration of annual NEE, typically by increasing ER relative to GPP (Baldocchi, 2008). In managed pastoral ecosystems, for instance, above-ground plant biomass is periodically removed by animal grazing and harvesting, temporarily interrupting plant production. In addition, trampling by animals can reduce effective leaf area and photosynthesis (Soussana et al., 2007). When green plant biomass is removed or damaged, photosynthesis is significantly reduced but soil respiration continues (Nieveen et al., 2005; Rogiers et al., 2005; Rutledge et al., 2014). For example, Nieveen et al. (2005) demonstrated that following a one-time grazing on peat soils there was net loss of CO_2 of about 32 gC m^{-2} over one month. Recovery of the plant canopy takes time, so that periodically cut or grazed pastoral systems are characterised by a saw-tooth pattern of NEE (e.g. Soussana et al., 2007; Wohlfahrt et al., 2008; Merbold et al., 2014). In Europe, biomass removal via harvesting or grazing often only occurs once or twice during the growing season (Lohila et al., 2004; Lloyd, 2006; Veenendaal et al., 2007; Rogiers et al., 2008), but sometimes up to six times for more intensive systems (Merbold et al., 2014). In contrast, year-round outdoor dairy grazing in New Zealand results in much more frequent grazing and consequent biomass removal (e.g. 8-15 times per year) (Luo et al., 2013; Rutledge et al., submitted). Since light response of NEE is often highly correlated with leaf area index (LAI) (Veenendaal et al., 2007), depending on the timing and amount of biomass removal by grazing or cutting, annual NEE and the net ecosystem C balance (NECB) can be severely affected (Lohila et al., 2004; Nieveen et al., 2005; Soussana et al., 2007; Veenendaal et al., 2007), although few studies have attempted to quantify the effect of grazing frequency and intensity on annual NEE (Lecain et al., 2000; Soussana et al., 2010; Kang et al., 2013).

Quantification of farm-scale exchanges of CO_2 is challenging, with most studies on agricultural peatlands using small chambers, where vegetation or bare soil are enclosed for short periods of time and CO_2 fluxes measured from rate of change of headspace CO_2 concentration (e.g. Lohila et al., 2003; Beetz et al., 2013; Leiber-Sauheitl et al., 2014). The main weakness of chamber techniques is that spatial variations in soil surface CO_2 fluxes lead to large uncertainties and the techniques themselves can lead to biases (Couwenberg, 2011; Gorres et al., 2014), and they are particularly unsuitable for investigating the impact of grazing events on NEE. In contrast, the eddy covariance (EC) technique can measure farm-scale CO_2 fluxes at timescales ranging from half-hour to annual, including disturbance events such as grazing or cultivation, but there have been relatively few studies that have applied EC to intensively farmed peatlands. For instance, Maljanen et al. (2010) reviewed more than 40 studies reporting GHG fluxes from agricultural peatlands in Nordic countries, but this review only included two studies that used the EC technique. While EC is well suited to ecosystem-scale studies, it is important to account for the dynamic flux footprint across a heterogeneous managed landscape, especially when plant biomass is changing relatively rapidly due to intensive grazing or harvesting. Also, continually changing plant biomass presents challenges for standard methods used to analyse the response of NEE to environmental drivers, for parameterising models necessary for gap-filling NEE to allow daily to annual sums to be compiled, and for comparing NEE across multiple sites. For instance, Merbold et al. (2014) showed enormous variability in the light response of GPP caused by repeated harvesting of pasture, and adopted a relatively complicated method for gap-filling that relied on careful recording of management interventions. Similarly, Veenendaal et al. (2007) made around 200 manual assessments of LAI per management intervention in order to separately calculate NEE light response parameters based on pasture growth stage. As yet there are no automated ways to account for this changing biomass but

Lohila et al. (2004) proposed calculating a “phytomass index”, based on the difference between night time and day time NEE, to account for seasonal crop development and biomass removal through harvesting.

In the Waikato region of New Zealand, where approximately one third of the country’s dairy production occurs, around 75,000 ha of former peat wetlands have been drained (Davoren et al., 1978), mostly for year round grazing by dairy cows. Ongoing subsidence of the peatland surface due to compaction and oxidation is a major concern (McLay et al., 1992; Pronger et al., in press). By measuring changes in peat mass through time, Schipper and McLeod (2002) estimated annual loss of farmed peatland soil C of 370 gC m^{-2} yr^{-1} during the first 40 years following initial development, which appears to be ongoing although at lower rates (Pronger et al., in press). In order to improve management of New Zealand agricultural peatlands, from both farm system sustainability and greenhouse gas accounting perspectives, knowledge of their net CO_2 balances is required. Towards this goal, improved understanding of the impact of key farm management practices, such as water table management, and grazing frequency and intensity, is necessary.

Our current understanding of CO_2 losses from agricultural peatlands in New Zealand is from a single year-long study on a dairy farm (Nieveen et al., 2005) and the spatial representativeness of this measurement is questionable. Alongside the original work of Nieveen et al. (2005), but not reported, NEE was also measured at three other dairy farms on peat during short campaigns using a mobile EC system in an attempt to determine the spatial consistency of results from the permanent site. However, one of the limitations of the work was how to account for differences in grazing management practices within and between farms which obscured meaningful comparisons. To compensate for this, we have now incorporated the phytomass index proposed by Lohila et al. (2004) to dynamically account for changing biomass while analysing NEE and deriving ecosystem model parameters. Our objectives here are to: (1) test the utility of the PI methodology for improving estimates of NEE in intensively managed farm systems with year-round rotational grazing on deep peat; (2) determine the spatial variability of NEE across dairy farms on peat, including on farms with different water table depths, in order to improve regional estimates of CO_2 emissions; and (3) to estimate the variability in annual NEE caused by differences in grazing timing and intensity.

4.2 Study site and methods

The Hamilton Basin, Waikato Region, North Island/Te Ika-a-Māui, New Zealand, contains several extensive peatlands up to 12 m deep and extending over a total area of 36,500 ha (Davoren et al., 1978) that developed within largely oligotrophic mires over the last 11–13,000 years on the surface of a broad alluvial fan deposited by former courses of the Waikato River (McCraw, 2011). Peat formation was dominated by vascular plants of the Southern Hemisphere family Restionaceae (Clarkson et al., 2004), but virtually none of this original vegetation remains. The peatlands were drained and developed for agriculture at various times over the past 40–100 years and have all suffered from extensive surface shrinkage, which is ongoing (Schipper and McLeod, 2002; Pronger et al., in press).

An eddy covariance tower was established on a Hamilton Basin dairy farm on 8-10 m depth of peat, at Rukuhia (37°50.88' S 175°14.06', altitude 47 m), and operated from 26-May-2002 to 25-May-2003 (Fig. 1). A mobile EC tower was deployed at three other dairy farms for periods of 3–4 weeks at a time between September 2002 and May 2003. These were: Komakorau, 26 km north-east of the Rukuhia site (Sept-Oct 2002 and May 2003); Reymer Farm, 1.5 km north-east (Jan-Feb 2003); and Moanatuatua, 15 km south-east (Feb-Mar 2003). In addition, the mobile tower was deployed 50 m from the permanent tower during an inter-comparison experiment that lasted 4 weeks during April–May 2003 (Fig. 1). The farms had a range of drain layouts and depths that resulted in different depths to the water table, especially during summer and autumn. All farms were rotationally grazed by dairy herds throughout the year, with pastures comprising mixtures of perennial ryegrass (Lolium perenne) and white clover (Trifolium repens).

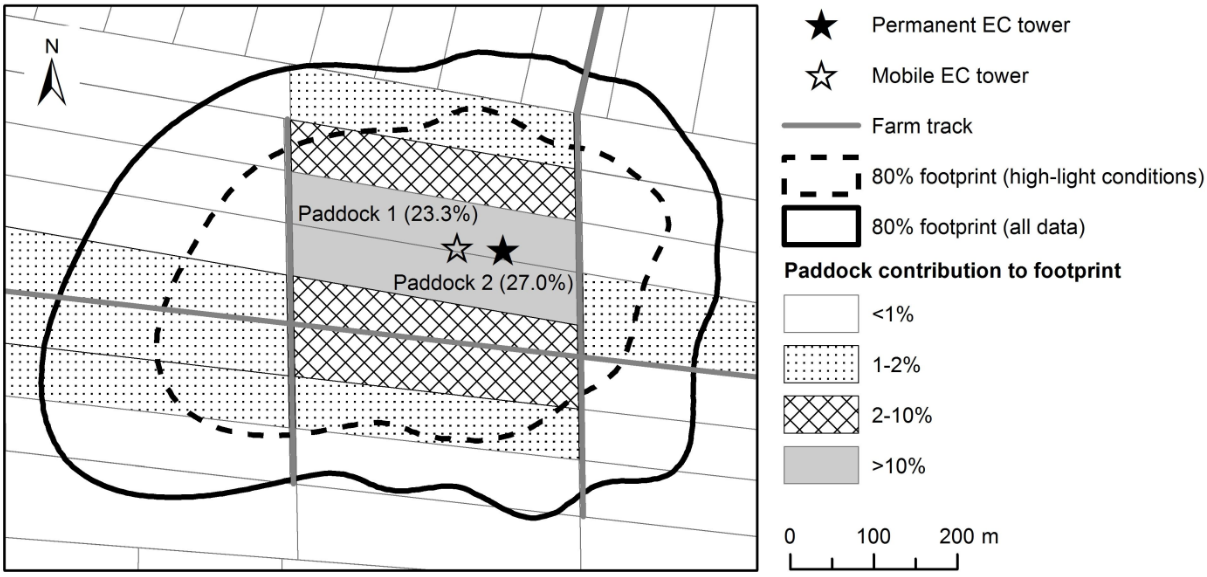


Figure 1 Map of the Rukuhia dairy farm site showing the location of the permanent and mobile EC towers in relation to field boundaries/drains and farm roads. Shading represents the calculated contribution of each paddock to the CO_2 flux measured by the permanent tower during the whole study period, while the contour lines contain the areas contributing 80% of NEE for all data (solid line) and 80% of daytime high-light NEE used to calculate the phytomass index (dashed line). For the two paddocks adjacent to the EC tower sites, percentages give their contribution to the overall flux footprint. Paddocks have areas of approximately 2 ha each.

Full details of the permanent EC system that was installed at the Rukuhia site are provided by Nieveen et al. (2005). Briefly, a sonic anemometer (CSAT3, Campbell Scientific Inc. (CSI), Logan, UT) and an air inlet were mounted at 4.3 m height on a 4.5 m tall steel-lattice tower close to the boundary of two paddocks each 350 m long and 60 m wide (Fig. 1). A closed path infrared gas analyser (IRGA) (LI-6262, LI-COR, Lincoln, NE) was housed in an insulated and temperature-controlled box at the base of the tower, with air drawn through a heated 5.2 m-long heated sample tube (4 mm internal diameter, Dekabon, Deane & Co., Glasgow, UK) at a flow rate of ~8 l min^{-1}. The IRGA was recalibrated at midnight every night by automatically shunting zero and span gases to the sample tube inlet. Raw signals from the sonic anemometer and IRGA were sampled at 10 Hz and stored on a laptop computer.

The mobile EC tower consisted of a CSAT3 sonic anemometer and an open path IRGA (LI-7500, LI-COR) mounted on a tower identical to the permanent system. The IRGA calibration was checked every 3-4 weeks using the same calibration gases as the closed path IRGA. Raw signals were collected by a datalogger (CR23X, CSI) and covariances calculated online.

For both EC systems, duplicate sets of environmental measurements were made using dataloggers (CR23X and CR10X, CSI), with most signals sampled at 10 s intervals and 30-minute statistics stored. Measurements included shortwave radiation and photosynthetically active radiation (PPFD) (LI-200X/LI-190S, LI-COR), air temperature and relative humidity at 4.3 m height (HMP-35C, Vaisala Inc., Helsinki, Finland), soil temperature at 0.1 m depth (107, CSI), rainfall (TB3, Hydrological Services, NSW), water table depth (SS3, Instrument Developments, Rangiora NZ), and atmospheric pressure (PCB220, Vaisala).

For the present research, we reprocessed all of the raw high frequency data from the permanent closed path EC system using EddyPro (version 5.1.1, LI-COR), whereas Nieveen et al. (2005) used a custom software package to calculate fluxes. Travel time within the sample tube was determined by maximising covariances between vertical wind speed and CO_2, signals were detrended using block averaging, and an analytic spectral correction was applied. A two-axis rotation was applied to correct for possible anemometer tilt. For the mobile EC system, raw covariances calculated by the datalogger were later adjusted for air density effects (Webb et al. 1980) and two-axis rotation. CO_2 fluxes were corrected for CO_2 storage changes in the air layer beneath the instruments, but these were generally very small.

Data quality checks included rejection of data when outside of plausibility bounds based on careful inspection of the data, lack of stationarity indicated by large standard deviations of CO_2 density, and for periods when either a sonic anemometer or the LI-7500 was affected by water droplets. During periods of under-developed turbulence, flux data were rejected if the friction velocity, u_*, was below a threshold value, u_{*thr} = 0.17 m s^{-1}, which we determined for the permanent EC tower by repeatedly calculating annual sums of NEE across a wide range of candidate u_{*thr} (0.02 – 0.3 m s^{-1}), and choosing the value above which the NEE sum remained relatively constant (Loescher et al., 2006). Remaining anomalous NEE were filtered by calculating statistics of each 30-minute time period across 30-day moving windows, flagging half-hours if their NEE fell outside of the range median±2 (daytime) or median±3 (night-time) standard deviations from the median for that half-hour band, and rejecting them if flagged multiple times as the window advanced one day at a time. It is highly likely that most short-term departures in NEE caused by cattle grazing in the EC footprint were rejected from the dataset.

We use the micrometeorological convention that positive NEE represents net CO_2 loss from the ecosystem and negative NEE represents net CO_2 uptake (Chapin et al., 2006).

To attribute measured NEE to the individual paddocks surrounding the permanent EC tower at Rukuhia, we used the analytical footprint model of Kormann and Meixner (2001). A 5 m grid was established, extending 1000 m in all directions from the tower, and the half-hourly flux contribution from each pixel was calculated. The average flux contribution from each of the paddocks in the main part of the footprint is shown in Fig. 1, which was dominated by the two paddocks immediately adjacent to the tower, together contributing 50% of the annual NEE.

Eddy covariance measurements are characterised by a large component of random noise at the half-hour timescale, and the effect of this on annual NEE sums should be quantified. In addition, the large amounts of data rejected during quality control procedures must be replaced by gap-filling procedures. To characterise the distribution of random measurement errors, we used the "successive days" approach of Hollinger and Richardson (2005), and evaluated the uncertainties introduced by measurement error and the gap-filling method following Dragoni et al. (2007) as described by Campbell et al. (2014). The effect of random measurement and gap-filling errors reduce as the timescale of summed NEE increases, whereas the effect of systematic errors in measurement and data processing systems is potentially much more serious. The choice of u_{*thr} for rejecting measurements during under-developed turbulence is a potentially large source of systematic error, so we quantified this separately as half the range of summed annual NEE calculated across a range of plausible u_{*thr} (0.12 – 0.22 m s^{-1}), which came to ±26 gC $m^{-2}yr^{-1}$.

Models of ecosystem CO_2 exchange are used to investigate the relative importance of different environmental drivers, for gap-filling time series to enable compilation of daily to annual sums, and for partitioning NEE into its components GPP and ER (Falge et al., 2001; Reichstein et al., 2005).

For light response of daytime NEE, a rectangular hyperbolic function is commonly used,

$$\mathrm{NEE} = -\left(\frac{\alpha.\mathrm{PPFD}.\mathrm{GP}_{\mathrm{max}}}{\alpha.\mathrm{PPFD} + \mathrm{GP}_{\mathrm{max}}} - \mathrm{ER}_0 \right) \qquad (1)$$

where α is the initial slope of the light response curve at low light (photosynthetic efficiency); GP_{max} is photosynthetic capacity or GPP at unlimited light level; and ER_0 is the y-intercept of the light-response function, equivalent to average ecosystem respiration at low light.

To account for the effects of seasonal plant development and biomass removal during harvesting on ecosystem light response, Lohila et al. (2004) used an empirical coefficient, the phytomass index, PI, calculated as the normalised value of daily ΔNEE_{N-D}, the difference between average night-time NEE and average daytime NEE during non-limiting light conditions. Lohila et al. (2004) normalised ΔNEE_{N-D} to unity at the peak of the growing season. Incorporated into Eq. 1,

$$\mathrm{NEE} = -\left(\mathrm{PI} \times \mathrm{VPDI} \times \frac{\alpha_{\mathrm{PI}}.\mathrm{PPFD}.\mathrm{GP}_{\max(\mathrm{PI})}}{\alpha_{\mathrm{PI}}.\mathrm{PPFD} + \mathrm{GP}_{\max(\mathrm{PI})}} - \mathrm{ER} \right) \qquad (2)$$

where VPDI is a scaling index based on measured saturation vapour pressure deficit, VPD, set to unity when VPD<0.5 kPa, and reducing linearly until reaching a value of 0.45 at VPD=2 kPa. Except for when PI=VPDI=1.0, the parameters α_{PI} and $GP_{max(PI)}$ have different physical meanings compared to Eq. 1 because they are both scaled by PI$\times$VPDI. Because the behaviour of PI is derived from measured ecosystem NEE response to environmental and plant growth conditions, it is likely to be a better predictor of primary production than intermittently harvested above-ground dry matter or measured leaf area index (Lohila et al., 2004).

To describe ER, the Lloyd and Taylor (1994) function is often used,

$$\mathrm{ER} = R_{10}.\mathrm{e}^{E_0\left(\frac{1}{283.15 - T_0} - \frac{1}{T - T_0}\right)} \qquad (3)$$

where T is air or soil temperature (K), E_0 is the effective activation energy, and R_{10} (μmol $m^{-2}s^{-1}$) is ER at reference temperature 10°C. T_0 and E_0 are usually fixed at 227.13 K and 308.56 K respectively (Lloyd and Taylor, 1994) and R_{10} fitted using non-linear regression. ER is the sum of plant dark respiration and below-ground heterotrophic respiration, both of which might be affected by the above-ground phytomass. Hence, we can include PI into Eq. 3,

$$\mathrm{ER} = \mathrm{PI} \times R_{10(\mathrm{PI})} . \mathrm{e}^{E_0\left(\frac{1}{283.15 - T_0} - \frac{1}{T - T_0}\right)} \qquad (4)$$

Models were fitted with non-linear regression using the Matlab Stats toolbox (Mathworks, MA)

To calculate PI year-round, we adapted the method described by Lohila et al. (2004). For each day, $\Delta \mathrm{NEE}_{\mathrm{N\text{-}D}}$ was calculated as the difference between mean night-time (PPFD<20 µmol $m^{-2}s^{-1}$) and mean non light-limiting daytime NEE (PPFD>700 µmol $m^{-2}s^{-1}$ except May to July where PPFD>500 µmol $m^{-2}s^{-1}$ was used – henceforth referred to as high-light conditions). At least 4 valid half-hour values of NEE were required at night, while the daytime minimum was 5 values. PI was then calculated as daily $\Delta \mathrm{NEE}_{\mathrm{N\text{-}D}}$ normalised by dividing by the maximum $\Delta \mathrm{NEE}_{\mathrm{N\text{-}D}}$ within non-overlapping 1-2 month windows. For days where $\Delta \mathrm{NEE}_{\mathrm{N\text{-}D}}$ could not be calculated because of low daytime light levels or missing data, we allowed the averaging periods for $\Delta \mathrm{NEE}_{\mathrm{N\text{-}D}}$ to extend up to one day either side (for some analyses, e.g. gap-filling NEE). If a gap still remained, the seasonal average PI was used instead.

The timing and intensity of grazing from one paddock to the next have components of chance associated with them, and we wanted to understand how much variation this factor might introduce to seasonal and annual NEE. For this, the effect of variations in grazing timing and intensity on NEE within a single paddock were modelled using a Monte Carlo simulation (1000 runs) of daily PI driven by grazing events that, for every model run, varied randomly in (a) the date of the year's first grazing event; (b) the minimum PI immediately following grazing; and (c) the PI value prior to the next grazing event. The "recovery rate" of PI was determined based on monthly standard grazing rotation length (days) for the Waikato region as incorporated in a whole farm dairy production model (Beukes et al., 2008), where the value applied was appropriate for the month the grazing event occurred. These rotation lengths varied from 20 days in spring to 100 days in early winter. Parameters for the model were adopted after careful inspection of patterns of daily PI from analysis of measured NEE. Once an annual sequence of daily PI was simulated, modelled NEE were calculated at half-hour time steps via Eqs. 2 and 4 (daytime NEE_{mod} and ER_{mod}, respectively) using measured driving variables (soil temperature and PPFD) and seasonal values of α_{PI}, $GP_{max(PI)}$ and $R_{10(PI)}$, then summed to seasonal and annual timescales.

4.3 Results

The Hamilton Basin experiences a warm-temperate climate, with relatively small seasonal differences in mean daily air temperature (T_{air}), and during the study period there were few days when maximum T_{air} exceeded 26°C or minimum T_{air} dropped below 0°C (Fig. 2a). Mean soil temperature (T_{soil}) was slightly warmer than T_{air} throughout the year. Over the 2002-3 study period, mean T_{air} (13.8°C) was close to the 1971-2000 normal (14.0°C). The seasonal range of maximum daily PPFD from winter to summer was 18–62 mol $m^{-2}day^{-1}$ (Fig. 2b).

Rainfall during the study year totalled 1199 mm, almost exactly the same as the long-term average 1190 mm (1971-2000) recorded at an official climate station, 4.6 km from the Rukuhia EC tower. Winter (June-July 2002) was slightly wetter than normal, as was December, while the spring period (August-October) was relatively dry. February and April 2003 received only 62% of normal rainfall, but the “drought” was interrupted by near-normal rainfall in March (Fig. 2c). The water table fluctuated within 0.5 m of the soil surface until January, and then declined to around 0.7 m depth from March through to May (Fig. 2c). Daily total NEE was positive (net CO_2 source) for most of the year, with only the spring months showing a consistent sink (Fig. 2d), coinciding with high light conditions and moderately cool T_{air} (Fig. 2a,b).

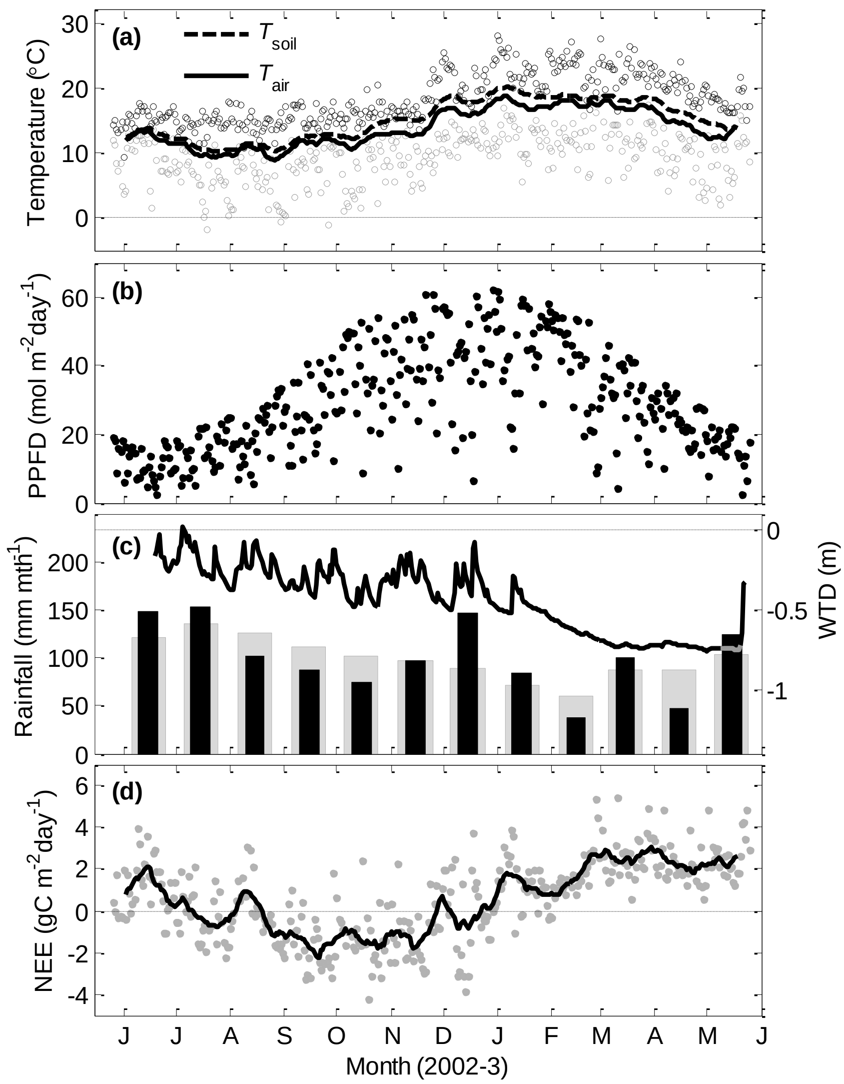


Figure 2 Variations in environmental conditions at the Rukuhia EC site, 26-May-2002 to 25-May-2003: (a) 15-day running means of air temperature, T_{air}, and 10 cm soil temperature, T_{soil} (lines), circles are T_{air} daily minimums (grey) and maximums (black); (b) daily total PPFD; (c) monthly total and long term average (1971-2000) rainfall (black and grey bars, respectively), and daily mean water table depth below the surface (WTD, line); and (d) daily total (points) NEE and 15-day running mean (line). Long-term rainfall data sourced from the National Climate Database (http://cliflo.niwa.co.nz/).

NEE displayed a saw-tooth pattern in response to removal of phytomass by grazing dairy cows, which was especially prominent in spring (Fig. 3a). Immediately following grazing, peak daytime NEE typically reduced by 50% or more, and then slowly recovered until the next grazing event. PI followed a similar pattern, with high values prior to grazing (0.8 – 1.0) and typically reducing to ~0.4 immediately after grazing (Fig. 3b). In early August, cows were break-fed (where cows get access to only a small part of the paddock each day) in the paddocks adjacent to the EC tower during wet soil conditions which led to severe trampling, and NEE was positive for several days until the pasture began to recover. PI during this time reached as low as 0.16. During the August-

September and October periods following grazing events, PI increased at a steady rate, reflecting pasture growth. The pattern was not always clear because the footprint of the EC measurements was affected by paddocks at a range of recovery stages, for example, mid-October to mid-November (Fig. 3b). For the period shown, the maximum footprint contributions from either of the paddocks adjacent to the EC tower ranged from 23–70% (Fig. 3c), with a mean of 46%. However, the footprint of daytime NEE used to calculate PI were reasonably well constrained to paddocks surrounding the EC tower (dashed contour, Fig. 1). The footprint at night was larger (not shown), and we found that night-time NEE were much less dependent on PI (see below), possibly because of being averaged across a larger area.

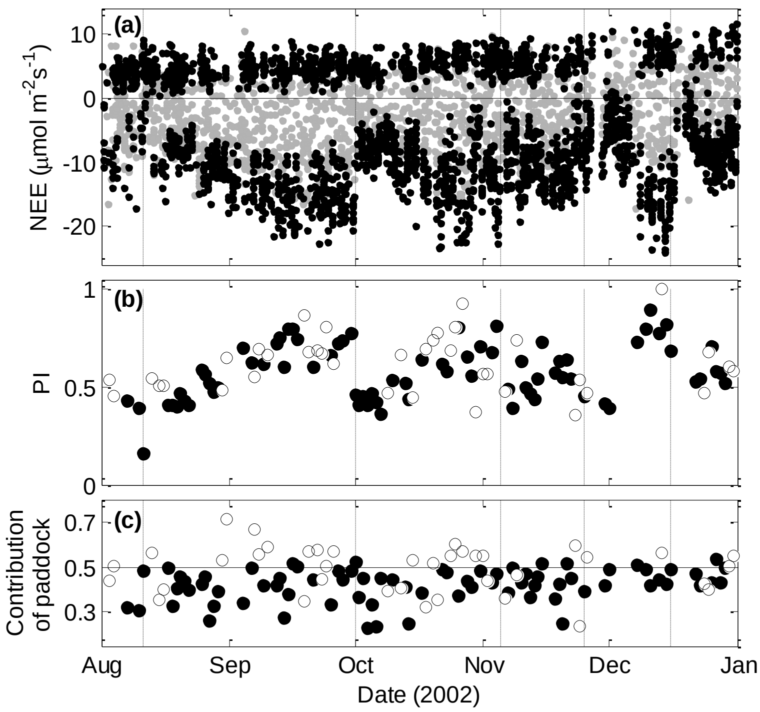


Figure 3 Net ecosystem CO_2 exchange (NEE) and phytomass index (PI) for grazed pasture at Rukuhia from late winter to early summer 2002: (a) time-series of 30-minute NEE, black points are night-time and high-light daytime values used to calculate PI; (b) daily values of PI, with open points showing days where the footprint was dominated by paddock 1, solid points paddock 2 (see map, Fig. 1); and (c) fractional contribution of dominant paddock NEE for each day, calculated using the footprint model. Vertical dashed lines indicate approximate timing of major grazing events affecting the EC footprint.

There was considerable variation in 30-minute daytime NEE at a given light intensity because of the large variation in phytomass caused by grazing cycles, as shown by a three month period in spring (Fig. 4a). This dataset was split into two parts, consisting of sets of alternate days. Both light response models were fitted to one set of data and validated against the independent NEE dataset. The standard model (Eq. 1) had relatively poor explanatory ability (R^2=0.64; RMSE=3.87), especially during moderate to high light conditions, and when there were positive NEE during daytime (Fig. 4b). The PI-modified model (Eq. 2) explained much more of the variation in observed NEE (R^2=0.85; RMSE=2.52) (Fig. 4c), including the positive NEE during daytime which occurred after the break-feed event in August that removed or trampled most of the phytomass.

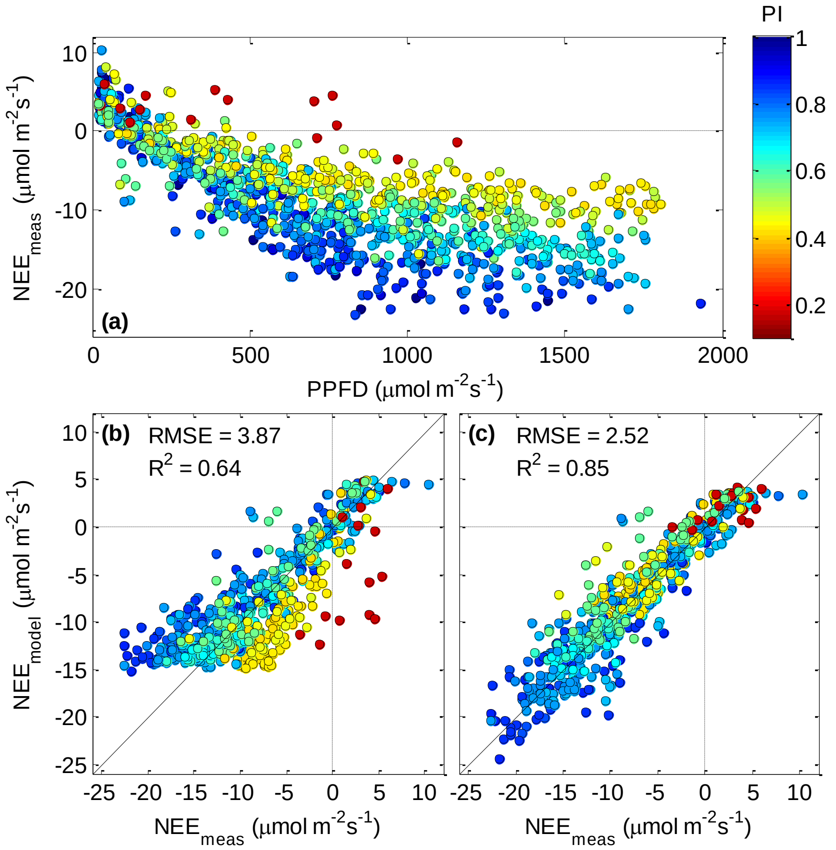


Figure 4 (a) Light response of measured daytime NEE (NEE_{meas}) to PPFD above grazed pasture on peat soil, for the period 01-August – 31-October-2002; and light response model validation fits for (b) standard light response (Eq. 1) and (c) light response incorporating the phytomass index (PI) and VPD index (Eq. 2). Model parameters for (b) and (c) were derived by fitting the models to 30-minute daytime (PPFD>20 mol $m^{-2}s^{-1}$) NEE for alternate days starting 02-August, model validation datasets shown were the other half of the dataset. All data points coloured according to PI, where high values indicate large pasture biomass.

The PI-modified light response model did not always perform well. During drying soil conditions in mid-summer and autumn, Eq. (2) failed to adequately predict positive NEE during low-light conditions by day, whereas Eq. (1) performed slightly better (Fig. 5b,c).

This likely occurred because ER was dominated by soil respiration from a deep aerobic zone in the peat, for which PI would have no logical explanatory ability. To account for this we separately fitted Eq. (1) and Eq. (2) to low- and high-light conditions, respectively ("dual fit" method), which led to improved fit statistics (Fig. 5d, R^2=0.83; RMSE=1.74). At low to moderate PI during January and February, the light response of NEE saturated at moderate light levels (Fig. 5a), with NEE significantly reduced compared to similar PI ranges in spring (Fig 4a).

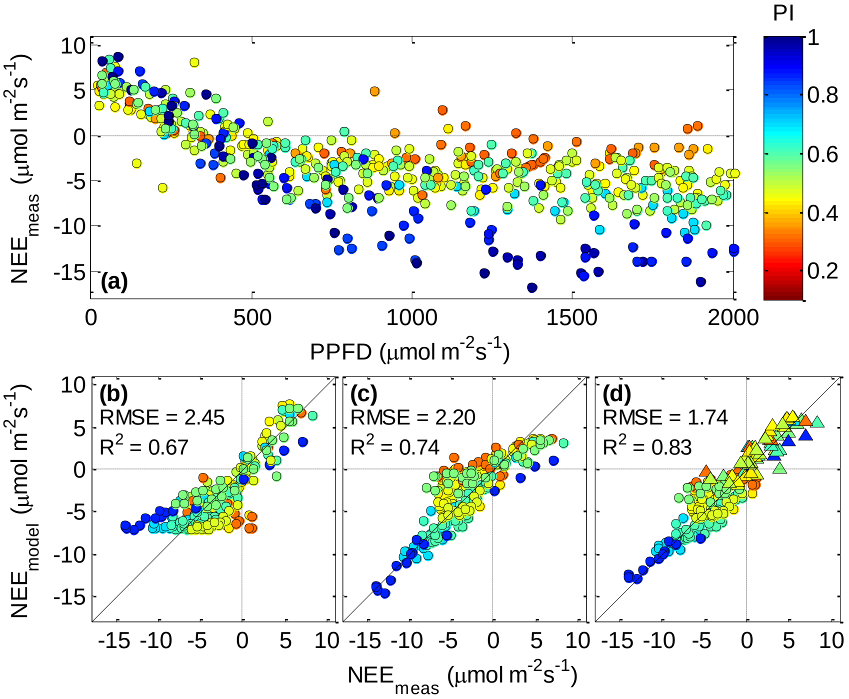


Figure 5 (a) Light response of daytime NEE to PPFD for the period 31-January to 28-February-2003; and model validation fits for (b) standard light response (Eq. 1); (c) light response incorporating the phytomass index (PI) and VPD index (Eq. 2); and (d) "dual fit" approach using Eq. 1 at low-light (triangles , PPFD<500 µmol $m^{-2}s^{-1}$) and Eq. 2 at high-light (circles). Model parameters for (b–d) were derived by fitting the models to 30-minute daytime (PPFD>20 µmol $m^{-2}s^{-1}$) NEE for alternate days starting 01-January, model validation datasets shown were the other half of the dataset. All data points coloured according to PI.

In theory, the contribution of plant dark respiration should also scale with the phytomass index (Lohila et al., 2004). We tested whether the PI-version of the Lloyd and Taylor equation (Eq. 4) explained more of the variation in night-time ER than the unmodified version (Eq. 3). Both equations performed relatively poorly (R^2 = 0.44 and 0.47; RMSE = 1.78 and 1.56 respectively), but overall the PI-version captured a more realistic range of ER. We attribute the lack of improvement by the PI-modified ER model to the differences between daytime and night-time flux footprints. The footprint of daytime high-light NEE was dominated by the adjacent paddocks, and 80% of these fluxes overall were derived from an area of 18.5 ha (about 9 paddocks, see dashed contour, Fig. 1). In contrast, 80% of night-time fluxes were sourced from an area of 97 ha (about 47 paddocks). Night-time fluxes were much more likely to represent an overall farm-scale average, explaining the lack of PI-sensitivity.

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Within a single farm, timing and intensity of grazing events will contribute significantly to the spatial (paddock to paddock) variability of NEE. Between farms, regional weather differences, stocking densities, water table management and sward types all likely add additional sources of variability. We compared NEE measurements from the permanent (Rukuhia) EC site with those from the mobile EC tower deployed for 3-4 week periods at three other dairy farms on peat. During one of these periods the mobile EC tower was deployed approximately 50 m from the permanent tower (Fig. 1). Figure 6 shows seasonal variation in mean night-time ER at the permanent site. Lowest ER occurred in July (3.3 ± 0.15 μmol $m^{-2}s^{-1}$) and increased through spring until mid-summer. A dropping water table and low rainfall led to reduced ER from February to May, except for March where ER was highest (9.2 ± 0.76 μmol $m^{-2}s^{-1}$), likely due to 100 mm rain in that month coupled with warm soil conditions (Fig. 2). Mean ER measured at the other farms were all very similar to those measured at the permanent site, even when the depth to water table differed substantially (Table 1). Daytime NEE were standardised by dividing by the daily PI determined at each site (NEE_{PI}) in order to account for the different timings and intensities of grazing at each farm as shown by differences in mean PI between paired sites (Table 1). At the Rukuhia site, daytime mean NEE_{PI} varied between –13.4 and –17 μmol $m^{-2}s^{-1}$ from June to January, but was only around –7 μmol $m^{-2}s^{-1}$ during the drier late summer and autumn periods (Fig. 6). The apparent lack of seasonality in mean NEE_{PI} from winter to early-summer arose because of similar photosynthetic capacity of the grass sward during high-light conditions. There was variation in NEE_{PI} from farm-to-farm, especially apparent when the mobile EC tower was at the Moanatuatua and Komakorau farms in March and May, respectively. For both of these pairings, the water table depth differed substantially between farms (Table 1). At Moanatuatua (with WTD > 1 m) the soil was evidently so dry that daytime GPP was highly constrained, and mean NEE_{PI} was indistinguishable from zero. Pasture growth at Rukuhia was clearly less constrained because the water table was 0.47 m higher. However, despite the substantially different water tables during the Moanatuatua-Rukuhia and Komakorau (K2)-Rukuhia pairings, mean night-time ER were almost identical between the sites.

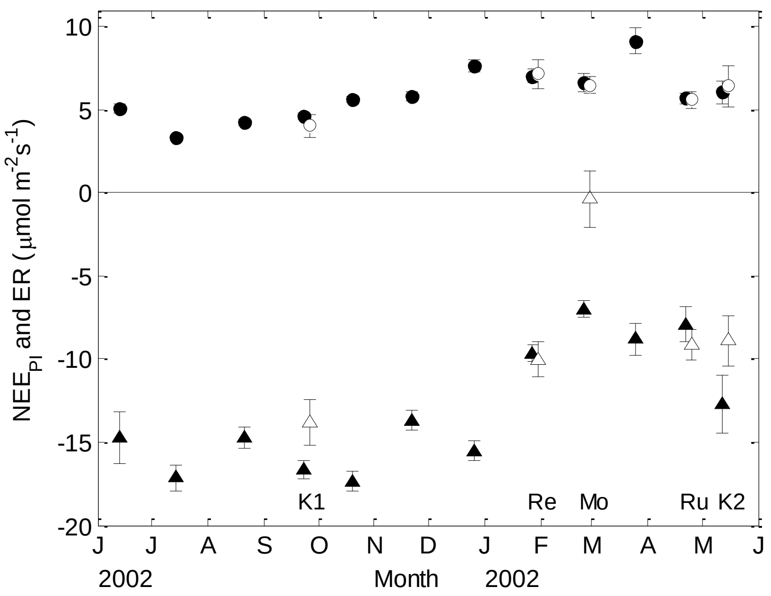

Figure 6 Mean night-time ecosystem respiration, ER (circles) and daytime NEE (high-light conditions) standardised by phytomass index (NEE_{PI}), for 3-4 week periods during the 2002-3 year. Solid points represent the permanent EC tower at Rukuhia and hollow points the mobile EC tower deployed at four sites (labels: K1 and K2-Komakorau; Re-Reymer; Mo-Moanatuatua; Ru-Rukuhia). Error bars show 95% confidence intervals around the means. Pairs of points slightly offset in time for clarity.

Table 1 Details of mobile EC tower deployment, with mean water table depth (WTD), night-time mean soil temperature (10 cm depth), and phytomass index (PI), with higher values indicating relatively high biomass. Values for Rukuhia permanent site are in parentheses. ND = no data.

Date range	Mobile EC site (site code)	WTD (m)	Ts (°C)	PI -
14-Sep – 05-Oct-2002	Komakorau (K1)	ND (–0.28)	14.1 (12.9)	0.67 (0.80)
15-Jan – 12-Feb-2003	Reymer (Re)	–0.62 (–0.53)	20.8 (19.1)	0.80 (0.58)
13-Feb – 12-Mar-2003	Moanatuatua (Mo)	–1.15 (–0.68)	18.1 (19.4)	0.42 (0.65)
11-Apr – 06-May-2003	Rukuhia (Ru)	–0.71 (–0.74)	16.2 (15.9)	0.85 (0.89)
07-May – 21-May 2003	Komakorau (K2)	–1.12 (–0.74)	14.7 (13.7)	0.99 (0.79)

To gap-fill Rukuhia NEE and partition it into its components GPP and ER, we used the PI-versions of the NEE light response and ER models (Eq. 2 and 4), with the models parameterised using NEE and driver variables (PPFD and soil temperature) for periods of approximately 2 months duration. During the dry late-summer and autumn of 2003, the “dual fit” light response model approach (Fig. 5) was adopted. ER consisted of measured values at night (PPFD<20 μmol $m^{-2}s^{-1}$) where they existed; otherwise the modelled values from Eq. 4 were used.

The effects of variations in grazing timing and intensity on NEE for a single paddock were modelled using the Monte Carlo simulation (1000 runs) of daily PI. From inspection of the PI data for the full year (e.g. Fig. 3) the minimum PI following grazing was set to vary randomly in the range 0.35 – 0.45 and PI prior to the next grazing in the range 0.85 – 1.0. To account for occasional trampling damage on wet soils, winter-time minimum PI was allowed to reach as low as 0.2.

Figure 7 shows seasonal sums of CO_2 exchange components along with means and 95% confidence bounds derived from our coupled grazing-NEE simulations. NEE was positive in all seasons except spring because ER mostly exceeded GPP. The largest CO_2 loss occurred in autumn (211.5 gC m^{-2} period^{-1}), mainly caused by reduced GPP. Simulated seasonal NEE agreed reasonably closely with measured NEE in most seasons, with seasonal variation attributable to grazing variations of 12–52 gC m^{-2}period^{-1}, with largest variation in spring and least in autumn.

Gap-filled annual CO_2 flux components for Rukuhia were NEE = 206 ± 12.2 gC m^{-2}, GPP = 1871 gC m^{-2} and ER = 2077 gC m^{-2} while the Monte Carlo simulation returned annual total NEE = 185 ± 30.2 gC m^{-2}.

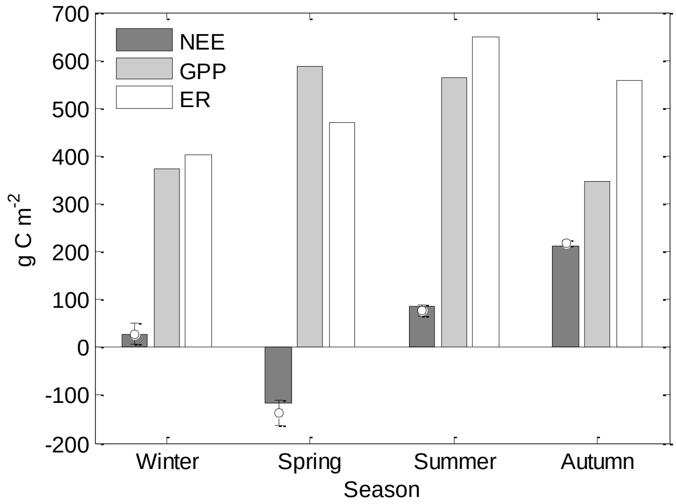

Figure 7 Seasonal total CO_2 flux components at the Rukuhia farm for the year 26-May-2002 to 25-May-2003, based on measured and gap-filled NEE. Circles and error bars show the seasonal sums and 95% confidence intervals of NEE derived from a Monte Carlo simulation of the grazing model.

4.4 Discussion

In contrast to the earlier analysis by Nieveen et al. (2005), who reported annual NEE indistinguishable from zero, we found CO_2 losses at Rukuhia of 206 ± 12.2 gC $m^{-2}yr^{-1}$ based on a re-analysis of the same dataset. This large discrepancy was likely caused by inadequate correction for high-frequency signal losses down the air sample tube in the original study which led to underestimates of both daytime and night-time NEE, but biased towards daytime net CO_2 uptake. In addition, our new analysis accounting for very large grazing-induced changes in pasture biomass using PI has also dramatically refined the gap-filling procedure. There have been several other studies of CO_2 budgets for boreal and temperate agricultural peatlands, with annual NEE ranging from –169 gC $m^{-2}yr^{-1}$ to 299 gC $m^{-2}yr^{-1}$ (Table 2). However, for intensively managed agricultural systems with grazing or harvesting, annual NEE of 116–299 gC $m^{-2}yr^{-1}$ (i.e. losses) are more typical, for which our revised NEE is mid-range. Annual GPP and ER from these other agricultural peatland studies are generally lower than our revised estimates (Table 2), probably because most of these sites have much shorter growing seasons than in New Zealand, with much colder winters. The only other warm-temperate peatland study was in California (Hatala et al., 2012) which reported relatively low GPP and ER from a degraded pasture (Table 2). Our ER estimate was similar to that reported for a Waikato dairy farm on mineral soils averaged across four years (2030 gC $m^{-2}yr^{-1}$), while our GPP was smaller than the four-year study (2196 gC $m^{-2}yr^{-1}$), but broadly similar to GPP in a drought year (1931 gC $m^{-2}yr^{-1}$) (Rutledge et al., submitted).

The flux of CO_2 to the atmosphere is only one contributor to farm-scale soil C losses. A full net ecosystem C budget requires knowledge of C imports (feed and manures) and exports (milk, meat, CH_4, and the drainage flux of dissolved C). Using farm production data we estimated the C content of exported milk solids to be 73.8 gC $m^{-2}yr^{-1}$. Using the average methane production rate from New Zealand dairy cows (Robertson and Waghorn, 2002; Laubach and Kelliher, 2004; Woodward et al., 2004), and Rukuhia farm stocking density (3.1 cows ha^{-1}), we estimated the C content of methane emissions from cows to be 30.2 gC $m^{-2}yr^{-1}$. There was no feed imported onto the farm, and we assume negligible meat exported. Currently we do not have data on dissolved organic C exports, so have assumed zero. Thus the NECB of the Rukuhia farm was approximately –310 gC $m^{-2}yr^{-1}$ (= –(206 + 30.2 + 73.8) gC $m^{-2}yr^{-1}$), which is 84% of average losses estimated for the first 40 years since a nearby peatland was developed for farming (Schipper and McLeod, 2002)

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Table 2 Annual CO_2 exchange components determined from eddy covariance measurements over agricultural peatlands. NECB is net ecosystem C balance calculated from NEE and other inputs and outputs of C (e.g. harvested grass or crops, milk solids, methane emissions). By convention, positive NEE represents CO_2-C loss from the ecosystem whereas negative NECB represents a loss of soil C (Chapin et al., 2006).

Land use, situation	Location (latitude)	Year	NEE	ER	GPP	NECB	Reference
			---------------gC m^{-2} yr^{-1}---------------				
Barley (cropped)	Finland (60°54′N)	2000-01	210			–336	Lohila et al. (2004)
Grass ley (mowed)	Finland (60°54′N)	2001-02	79			–452	
Meadow, (managed water table)	UK (51°12′N)	2002	–169			–59	Lloyd (2006)
Meadow (mowed)	Netherlands (52°01′N)	2004-05	–6	1539	1545	–424	Veenendaal et al. (2007)
Dairy (mow/graze pasture)	Netherlands (52°02′N)	2004-05	134	1597	1463	–423	
Pasture/hay (alpine)	Switzerland (47°03′N)	2002-03	120	1321	1201		Rogiers et al. (2008)
		2003-04	256	1614	1358		
		2004-05	116	1371	1253		
		2003	172	1619	1447	–373[1]	
Pasture (degraded)	California (38°02′N)	2009-10	299[2]	1493	1182		Hatala et al. (2012)
		2010-11	174[2]	1765	1557		
Dairy (rotational grazing)	New Zealand (37°51′S)	2002-3	206	2077	1871	–310	This study

Notes:

[1]Mean of two estimates

[2]Reported NEE inconsistent with GPP and ER.

The ability to determine CO_2 fluxes from the large areas of farmed peat soils around the world is important for developing regional and national scale estimates of greenhouse gas emissions (Couwenberg, 2011). While the best approach would clearly be to make multiple annual large scale measurements, these can be limited by available resources (both equipment and time). One of our objectives was to determine whether a permanent EC site paired with a mobile EC tower might be a useful and less resource-intensive approach for predicting larger scale fluxes of CO_2. Towards this objective, use of the PI methodology made it possible to directly compare NEE measurements at pairs of farms despite different timings of grazing cycles. Across the five direct comparisons between the permanent and mobile tower sites during three seasons and a range of water table depths, there was close agreement for night-time respiration rates but less agreement for daytime NEE even when corrected for differences in phytomass. The weaker agreement in daytime NEE suggested there are factors other than plant biomass, such as water table depth and soil moisture content, that need to be taken into account when trying to compare CO_2 exchange between different farms. The largest differences in daytime NEE between the paired farms occurred when the mobile tower was at deeply drained sites in autumn – Komakorau and Moanatuatua – where the water table was up to 0.47 m lower than at the permanent site. At Moanatuatua there was essentially no net CO_2 fixation whereas the permanent site continued to fix CO_2 (Fig. 6). This large difference in daytime NEE_{PI} between sites during dry periods was due to differences in GPP because night-time ER were almost identical. The decline in GPP was likely due to water limitation on plants. When water tables in peat decline, water availability to plants depends on pore structures and remaining soil water may be inaccessible to plants. For example, McLay et al. (1992) found that Waikato agricultural peats retained large amounts of soil water at high suctions (>1500 kPa), leading to increased drought sensitivity and plant stress compared to mineral soils.

Somewhat surprisingly, night-time ER at site pairs were similar even when WTD differed by as much as 0.47 m, whereas we expected that deeper water tables would have led to greater respiration. However, Mäkiranta et al. (2009) showed that drying of surface peat layers begins to limit respiration rates even at moderate WTD (0.3 – 0.6 m), because the effect of moisture stress on microbial communities near the peat surface is not compensated for by the increased thickness of the aerobic layer as the water table continues to fall. In support of this hypothesised moisture limitation on respiration from near-surface peat, we observed increased ER at Rukuhia in March (Fig. 6) following rainfall that did not lead to water table recovery (Fig. 2c), suggesting that surface soil moisture conditions became more favourable to microbial decomposition. More broadly, Couwenberg et al. (2010) compiled data from a number of studies that demonstrated the control on NEE by WTD in peat soils, with net CO_2 losses rapidly increasing as WTD dropped to 0.4 m beneath the surface, but stabilising at deeper WTD. An important management implication is that maintaining a higher water table would likely decrease

CO_2 losses, mainly by increasing pasture growth. Consequently, identifying an optimum WTD may provide both environmental and production benefits.

There was close agreement in NEE between farms when water table depth and temperatures were similar (e.g. the Reymer pairing) with NEE_{PI} and night-time respiration rates indistinguishable between sites, despite large differences in PI (Table 1). This suggested that CO_2 fluxes from Rukuhia could be extrapolated to other similar peat farms in the Waikato region. To extrapolate more broadly to other sites will require a better characterisation of other controlling factors, including peat type and broader farm management regimes.

Incorporation of the phytomass index into a standard light-response function dramatically improved our ability to describe the impacts of grazing as a key factor responsible for driving variability in NEE. This is in contrast to previous studies in agricultural ecosystems which have not been able to automatically discriminate widely varying daytime NEE caused by grazing or harvesting (Rogiers et al., 2005; Merbold et al., 2014; Rutledge et al., submitted). Automated PI analysis also allowed us to gap-fill NEE without reliance on detailed farm records across multiple paddocks within the EC footprint. In contrast, Merbold et al. (2014) subdivided periods between harvesting events and used look-up tables to describe daytime NEE for similar stages of grassland canopy development or management intervention. Similarly, Veenendaal et al. (2007) separated out periods with grazing or cutting and fitted light response parameters depending on pasture growth stage. We were also able to compare NEE measured at pairs of peatland farms by standardising daytime NEE by PI to account for different grazing cycles between farms.

Further work remains to confirm that PI is an effective measure of standing pasture biomass in response to year-round rotational grazing cycles. Encouragingly, Lohila et al. (2004) showed that PI derived from NEE closely followed the seasonal course of LAI for barley and grass crops, including recovery from grass cutting. For rotational grazing systems, such validation can only be done in situations where the EC footprint is well matched to the spatial and temporal scale of land treatments. Coupling PI analysis to spatial attribution of EC footprint, as demonstrated in Fig. 1, is a powerful methodology towards this goal. Regardless of the need for finer-scale validation, automated PI analysis allows wider use of the EC technique to develop regional CO_2 (and potentially other grazing-dependent trace gases such as N_2O) budgets without reliance on resource-intensive biomass measurements or compilation of detailed farm records.

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4.5 Conclusions

During a year of more-or-less average climate conditions, we measured net CO_2 loss of 206 gC $m^{-2}yr^{-1}$, for a rotationally grazed dairy farm on deep peat in New Zealand, which was consistent with findings from a number of studies in other parts of the world. Proper estimation of this loss was critically dependent on the use of the phytomass index to account for large variations in above-ground biomass caused by grazing cycles. Based on a limited number of short-term comparisons, the NEE measured at our one permanent site appeared to be reasonably representative of other farms with similar water table depths but much less so for sites with deep water tables during summer and autumn. Night-time ecosystem respiration rates were very similar between sites, even when the depth to water table differed substantially. In contrast, daytime NEE (adjusted for differences in phytomass) were markedly different at sites with excessively deep water tables, most likely due to water stress impairing pasture growth. Surprisingly then, deep water table control on CO_2 losses from peat was not via increased respiration, but via reduced photosynthesis. This finding suggests that managing water tables to be closer to the peat surface will both improve plant production and reduce net CO_2 losses.

Including other C exports from the Rukuhia farm, we estimated that the farm system lost around 310 gC $m^{-2}yr^{-1}$, which is ~84% of average losses estimated for the first 40 years since a nearby peatland was developed for farming (Schipper and McLeod, 2002). This suggests that losses are ongoing under present-day rotational grazing management for dairy farming.

4.6 Acknowledgements

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