Wall, A.M., Campbell, D.I., Goodrich, J.P., and Schipper, L.A. 2022. Continuous paddock-scale GHG measurements from grazed Waikato pastures on mineral and organic soils. In: *Adaptive Strategies for Future Farming*. (Eds. C.L. Christensen, D.J. Horne and R. Singh). http://flrc.massey.ac.nz/publications.html. Occasional Report No. 34. Farmed Landscapes Research Centre, Massey University, Palmerston North, New Zealand. 10 pages.

CONTINUOUS PADDOCK-SCALE GHG MEASUREMENTS FROM GRAZED WAIKATO PASTURES ON MINERAL AND ORGANIC SOILS

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Abstract

Greenhouse gas (GHG) emissions from the agricultural sector account for almost 50% of New Zealand's gross emissions. For inventory purposes, these emissions are estimated by combining activity data with relevant emission factors. However, recent technological advances now allow for the measurement of continuous GHG emissions at paddock scale (i.e. several hectares) for individual locations and/or treatments.

Between 2017 and 2018, we measured GHG emissions at two commercial Waikato dairy farms with one each on organic (drained peatland) and mineral soils. Our objectives were to (1) quantify GHG emissions from grazed pastures, and (2) test the effect of management practices such as pasture renewal and inclusion of plantain in the pasture sward on GHG emissions using a novel split-footprint methodology. Continuous measurement of gaseous CO_2, N_2O and CH_4 fluxes was made using the eddy covariance technique and, when coupled to measurements and estimates of all other flows of carbon (C), allowed for the calculation of the net ecosystem carbon balance (NECB) and GHG balance. Here, we report one year of data for the organic soil, and four years for the mineral soil. Utilising the split-footprint methodology, GHG balances were calculated separately for two adjacent paddocks in years 2-4 at the mineral soil site resulting in eight site-years of data.

We found that annual GHG emissions ranged from 13.9 to 22.4 t CO_2-eq ha^{-1} y^{-1}. Accounting for change in C stored in the ecosystem (i.e. that stored in the vegetation and soil, and calculated as NECB) was important for determining annual GHG emissions and contributed ~40% to our measurements. Moreover, the inclusion of the change in ecosystem C storage is essential for accurate estimation of GHG emissions from organic soils, but maybe less important for long-term mineral soil emissions where the average C balance from grazed pastures is near zero. Management practices such as pasture renewal and day-to-day management of the pastures can influence GHG emissions with pasture renewal activity emitting an additional 3 t CO_2-eq ha^{-1} relative to a non-renewed pasture. We also tested the hypothesis that the inclusion of plantain in the pasture swards can decrease N_2O emissions and thus GHG emissions, but poor establishment and retention of the plantain resulted in inconclusive results. Using new technology and innovative methodology, we were able to make some of the first paddock-scale GHG emission measurements in New Zealand while testing the effects of management practices on real-world commercial farms.

Introduction

Agriculture is the largest contributor to New Zealand's gross greenhouse gas (GHG) emissions accounting for 48% of emissions (39,617.7 kt CO_2-eq) in 2019 (MfE, 2021). Methane (CH_4) and nitrous oxide (N_2O) contribute 77% and 20% to agricultural emissions respectively. Agricultural emissions reported within New Zealand's inventory are mostly attributed to the categories of enteric fermentation (CH_4), manure management (CH_4 and N_2O), and agricultural soils (N_2O). Currently, changes in soil carbon (C) from mineral soils are assumed net zero, and not included. However, managed organic soils including drained peatlands have long-term continuous carbon dioxide (CO_2) and N_2O emissions caused by the mineralisation of ancient organic matter, so these are accounted for separately. Carbon losses are accounted for in the LULUCF sector while N_2O is included in the Agriculture sector. There has been little research on emission factors for NZ organic soils, so IPCC default values are currently used.

GHG emissions reported in New Zealand's inventory are calculated using a combination of activity data and emission factors. Similarly, individual farmers can calculate emission profiles for their farms using available tools such as OverseerFM, Farmax, and several others (AgMatters, 2022). Although not practical to apply to all farms, technology such as the eddy covariance (EC) method (Baldocchi, 2014) now allows for continuous, direct measurements of gaseous GHG emissions (i.e. CH_4, N_2O and CO_2) from agricultural pastures at paddock-scale. To determine the true GHG emissions of grazed pastures, estimates of the change in C stored in the ecosystem (including the soil) and enteric CH_4 emissions from the ruminant animals are also required. Estimation of the change in ecosystem C storage can be made using the net ecosystem carbon balance (NECB) approach (Chapin *et al.*, 2006) which combines the gaseous CO_2 emissions with non-gaseous flows of C into and out of the pasture ecosystem. Finally, the calculation of enteric CH_4 emissions can be made using grazing feed intake information coupled with known emission factors, thus completing the terms required to calculate the total GHG emissions from the pasture system.

Between 2017 and 2021, we used the EC and NECB measurement techniques to quantify GHG emissions from two Waikato dairy pastures, one each on mineral and organic soils. The objectives of these measurements were two-fold: firstly, to establish what the total GHG emissions from these pastures were, and secondly to test the impact of management practices on GHG emissions. Using a novel split-footprint methodology (Wall *et al.*, 2020; Goodrich *et al.*, 2021), the management practices of pasture renewal and inclusion of plantain in the pasture sward were investigated.

Methods

Site Description

Measurements were made on two commercial Waikato dairy farms. The first was on mineral soil (Mottled Orthic Allophanic soil (Hewitt, 2010)) near Waharoa in the Waikato. The farm was 199 ha in size with a stocking rate of ~2.5 cows ha^{-1}. Measurements were made between 2017 and 2021 across a two-paddock site of approximately 6 ha in size. In March 2018, one of the two paddocks (paddock B) was cultivated (single herbicide application followed by direct drilling four days later) with a sward containing plantain (*Plantago lanceolata*), ryegrass (*Lolium perenne*) and clover (*Trifolium repens*). The second paddock (paddock A) was an established ryegrass/clover sward typical of the area that remained unchanged through the duration of the measurements.

The second measurement site was an organic soil within the Moanatuatua peatland near Hamilton. The wider Moanatuatua peatland was drained between the 1930s and 1980s and is primarily used for dairy farming and supplemental feed cropping, with some blueberry orchards (Campbell *et al.*, 2021). A field site was established in 2018 covering 7.8 ha across two paddocks of ryegrass/clover pasture, on peat 6-8 m deep. The dairy farm covered a 329 ha area of peatland drained in 1975 and had a stocking rate of 2.4 cows ha^{-1}. To date, only the first year of measurements has been analysed and presented here.

Eddy covariance measurements

Measurements of gaseous GHG fluxes of CO_2, CH_4 and N_2O were made using the EC technique (Baldocchi, 2014). Equipment at both sites was similar with detailed descriptions of the systems and data processing provided by Liang *et al.* (2018) and Campbell *et al.* (2021) for the mineral and organic sites respectively. Briefly, 3-dimensional wind data obtained from sonic anemometers (CSAT3/CSAT3B; Campbell Scientific Instruments, Logan, Ut, USA) was combined with trace gas measurements from an infrared gas analyser for CO_2 (LI-7200/LI-7500RS; LI-Cor Inc, Lincoln, NB, USA) and a continuous wave quantum cascade laser for CH_4 and N_2O (Aerodyne Research Inc., Billerica, MA, USA). Data were collected at 10 Hz and fluxes were calculated at 30-minute intervals. The 30-minute fluxes underwent rigorous quality control followed by gap-filling using machine learning techniques to produce continuous datasets (for more details see: Wall *et al.*, 2020; Goodrich *et al.*, 2021). During years 2-4 at the mineral soil site, the paddock orientations were modified to allow the gaseous fluxes to be calculated for each of the two paddocks separately (see: Goodrich *et al.*, 2021). Therefore, each year yield two datasets of gaseous flux data and subsequent GHG emissions.

GHG calculation

EC measurements provided the gaseous flux data, while measurements and farm records allowed for the calculation of the NECB. Non-gaseous components of the NECB included the C imported to the paddock in supplemental feed and fertiliser, the removal of C in animal intake of pasture and supplemental feed, and leaching losses. The quantity of C returned to the paddocks in excreta deposited by the grazing cows was estimated from the quantity of feed eaten, the digestibility of the feed and grazing duration. Detailed descriptions of these calculations and methodology can be found in Wall *et al.* (2020).

Because EC measurements often do not accurately capture cow-specific gaseous fluxes (such as CO_2 respired and enteric CH_4 emissions (Kirschbaum *et al.*, 2020)), enteric CH_4 emissions were calculated as feed eaten (pasture and supplemental) within the measurement areas multiplied by the emission factor of 21.6 g CH_4 kg DM^{-1} (MfE, 2021). Accordingly, two CH_4 contributions are included in the GHG emissions calculation: (1) the atmosphere-ecosystem CH_4 exchange as captured by the EC system excluding the data obtained during the presence of the cows, and (2) calculated enteric CH_4 emissions.

Finally, the GHG emissions were calculated as:

$$NGHGB = \frac{m_{\mathrm{CO2}}}{m_C} \times \left[\mathrm{NECB} - F_{\mathrm{CH4,surface}} - \left(\frac{m_{\mathrm{C}}}{m_{\mathrm{CH4}}} \times F_{\mathrm{CH4,enteric}}\right)\right] + \gamma_{\mathrm{CH4}} \times \left[\left(\frac{m_{\mathrm{CH4}}}{m_{\mathrm{C}}} \times F_{\mathrm{CH4,surface}}\right) + F_{\mathrm{CH4,enteric}}\right] + \gamma_{\mathrm{N2O}} \times F_{\mathrm{N2O}}$$

Where: NGHGB is the resultant GHG emissions; m_{CO2}, m_{CH4} and m_C are the molar masses of CO_2 (44), CH_4 (16) and C (12) respectively; NECB is the net ecosystem carbon balance (in units of t C ha^{-1}); $F_{CH4,surface}$ and $F_{CH4,enteric}$ are the CH_4 flux as measured by the EC system and calculated enteric CH_4 emissions respectively (both in t CH_4 ha^{-1}); and F_{N2O} is the flux of N_2O

(as t N_2O ha^{-1}). Finally, global warming potentials for CH_4 (γ_{CH4}) of 27.2 and N_2O (γ_{N2O}) of 273 (IPCC, 2021) were used to convert all data to units of t CO_2-eq ha^{-1}. The data presented here for years 3 and 4 of the mineral soil is preliminary.

Results and Discussion

GHG Emissions

Measured annual GHG emissions from the two dairy pastures ranged from 13.9 to 22.4 t CO_2-eq ha^{-1} y^{-1} from eight site-years of data (Figure 1). NECB data for all site-years reported a loss of ecosystem C that ranged from 4.2 to 13.5 t CO_2-eq ha^{-1} y^{-1}. As a proportion of the total GHG emissions, ecosystem C loss (as CO_2) contributed between 26% and 60% annually at our measurement sites. N_2O and the CH_4 emissions from the soil and excreta ($F_{CH4,surface}$) were much more conservative. N_2O emissions from the mineral soil ranged from 1.7 to 3.1 t CO_2-eq ha^{-1} y^{-1}, while the one year of data from the organic soil had an emission of 1.7 t CO_2-eq ha^{-1} y^{-1} and was similar to those of years 3 and 4 from the mineral soil. On average N_2O emissions contributed 13% to total GHG emissions. CH_4 emissions from the soil and excreta ($F_{CH4,surface}$) were consistently between 0.9 and 1.4 t CO_2-eq ha^{-1} y^{-1} from the mineral soil, but only 0.5 t CO_2-eq ha^{-1} y^{-1} from the single year of organic soil data. Analysis of the additional years of the surface CH_4 emissions from the organic soil will help establish if this pattern is consistent. Finally, calculated enteric CH_4 emissions ($F_{CH4,enteric}$) varied from 5.3 to 7.3 t CO_2-eq ha^{-1} y^{-1} and were dependent primarily on annual pasture growth.

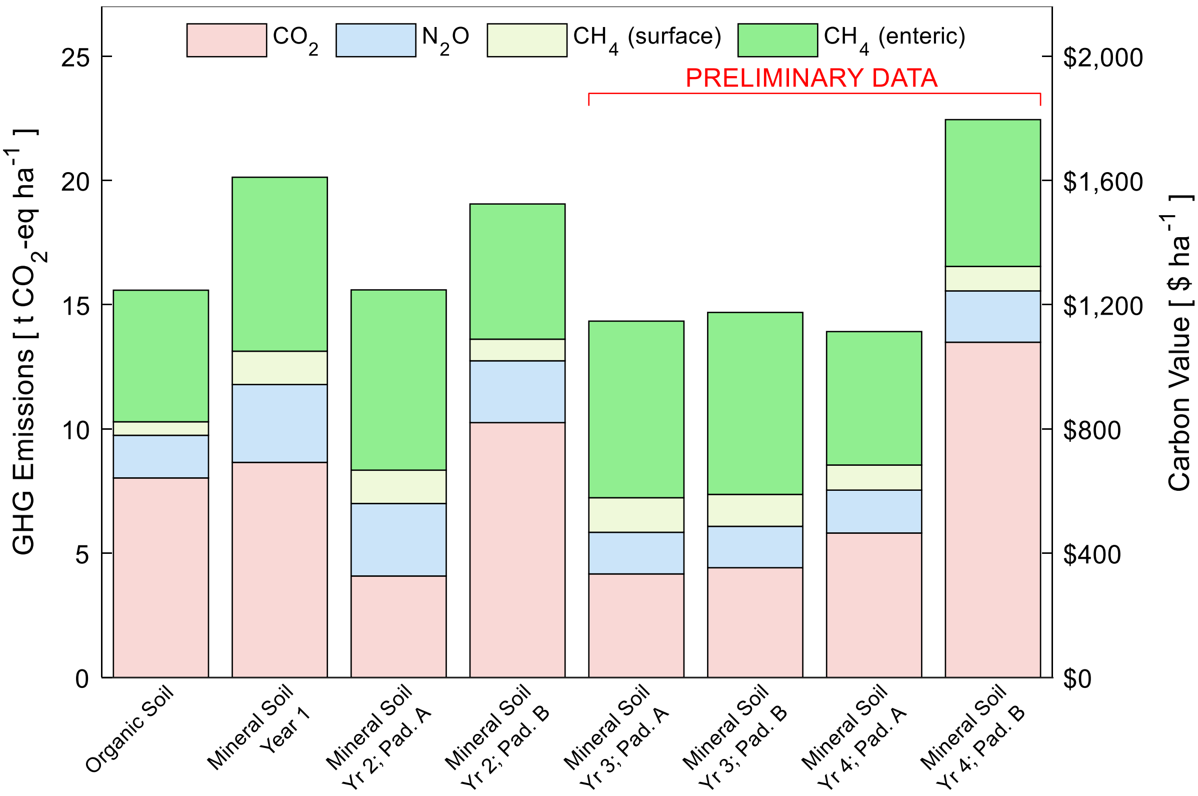

Figure 1: Greenhouse gas emissions from organic and mineral (calculated individually for paddocks A and B in years 2 to 4) soils separated into contributions by each gas. The right y-axis illustrates the economic value of these emissions at a carbon price of \$80 per t CO_2-eq. The emissions reported as CO_2 are determined from the change in ecosystem C storage. Note that years 3 and 4 for the mineral soil are preliminary data.

GHG emissions from the eight site-years of data illustrate the importance of the change in ecosystem (or soil) C to the total GHG emissions. For example, averaging across all data, N_2O and total CH_4 emissions were 2.2 and 7.5 t CO_2-eq ha^{-1} y^{-1} respectively, while the average C loss was 7.4 t CO_2-eq ha^{-1} y^{-1} accounting for ~40% of total emissions. While such C loss would not be expected to occur continuously and at all locations (see next section) it does highlight the contribution soil C can have to total emissions, especially at the annual time scale.

Influence of soil C

The importance of changes in ecosystem (or soil) C to individual years GHG emissions have been noted above but requires further exploration as to whether similar C losses could be expected elsewhere. Analysis of all available C balance data (Figure 2) indicates that organic and mineral soils need to be considered separately. Although only very limited data for organic soils is available to date, the C balance obtained for our one year of data was at the low end for the C losses reported in the literature. Continuous loss of C from organic soils drained for agriculture is not unexpected given the ongoing subsidence of these soils reported elsewhere (Pronger *et al.*, 2014). Both previous studies reporting C balances from organic soils in New Zealand (Campbell *et al.*, 2015; Campbell *et al.*, 2021) found C losses, and one of those studies (Campbell *et al.*, 2021) report a C loss of 31 t CO_2-eq ha^{-1} y^{-1} for the same year at a nearby site (2.7 km away). Assuming N_2O and CH_4 emissions were to be similar to those reported here, grazed pastures on organic soils could be emitting up to 40 t CO_2-eq ha^{-1} y^{-1}, which would have an economic value of $3200 per hectare (at a carbon price of $80 t CO_2-eq).

In contrast to organic soils, evidence from >40 site-years of data suggests that mineral soils on grazed pastures have an average C balance of near-zero (Wall *et al.*, 2021). Soil C stock measurements provide further support that mineral soils under continuous pasture are near steady-state (Schipper *et al.*, 2014). While the overall average was near zero, individual years experienced gains or losses of C. If mineral soils under grazed pastures are indeed at a long-term steady-state for ecosystem C change, the average GHG emissions would be solely due to CH_4 and N_2O. For our site, that would represent an average emission of 9.7 t CO_2-eq ha^{-1} y^{-1}.

Changes in the ecosystem, or soil, C likely play a different role in determining the GHG emissions of grazed pastures depending on soil type and measurement duration. For organic soils, the inclusion of changes in ecosystem C is essential to accurately estimate GHG emissions. Similarly, quantification of annual GHG emissions, regardless of soil type, requires a measure of ecosystem C change, which may enhance or offset N_2O and CH_4 emissions. However, over longer time scales (e.g. decadal), ecosystem C contributions to GHG emissions from mineral soils may be able to be ignored, similar to the approach currently taken by the New Zealand GHG Inventory (MfE, 2021). Finally, while C loss from drained organic soils is included in the New Zealand GHG Inventory (albeit within the LULUCF category), more research is needed to obtain New Zealand specific emission factors allowing for a more accurate estimation of GHG emissions from these soils.

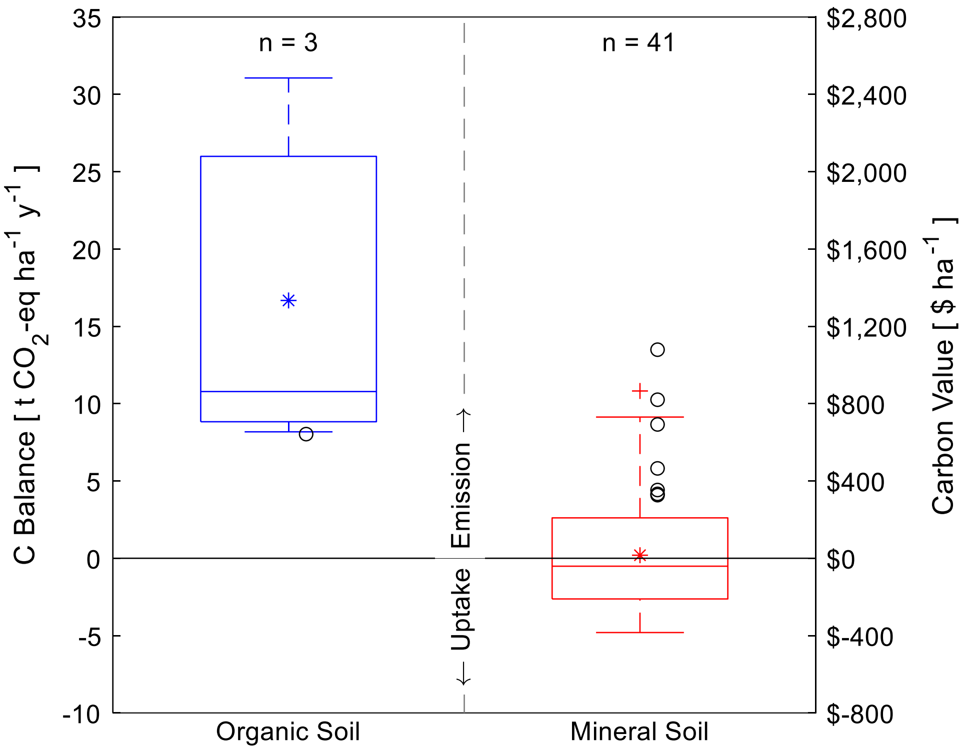

Figure 2: Summary of all available carbon (C) balance data from New Zealand dairy pastures. The asterisk indicates the mean C balance and the open circles represent data presented in this research. The right y-axis illustrates the economic value of these emissions at a carbon price of $80 per t CO_2-eq. Data includes all published and University of Waikato unpublished data, noting that the preliminary year 3 and 4 data from the mineral are not included in the mineral soil boxplot. (Adapted from Wall *et al.*, 2021)

Pasture renewal

The two paddocks within the mineral soil site were also used to test the effects of management practices on GHG emissions. One of the paddocks underwent pasture renewal (paddock B; referred to as renewed pasture), while the other remained as control (paddock A; referred to as continuous pasture). The pasture renewal began on the 10th March 2018 with herbicide application, followed by seeding via direct drilling on 14th March, and the first grazing was on 15th May. During the period between herbicide application and the first grazing (referred to as the "renewal period"), GHG emissions from the renewed paddock were around 10 t CO_2-eq ha^{-1} compared with 7 t CO_2-eq ha^{-1} from the continuous pasture paddock (Figure 3). Increased emissions from the renewed paddock were due to increased N_2O release, and net CO_2 emission due to the absence of, and then reduced photosynthetic uptake as the existing sward died, and the new sward began to grow. The absence of grazing during renewal precluded any enteric CH_4 emission over this renewal period. For the continuous pasture paddock, a grazing event during the renewal period was the major source of GHG emissions as the cows consumed much of C stored in the pasture as they grazed, emitting it as respired CO_2 and enteric CH_4 with a portion also exported in milk. Consequently, the difference in GHG emissions between the two paddocks was much smaller than had the grazing event not occurred.

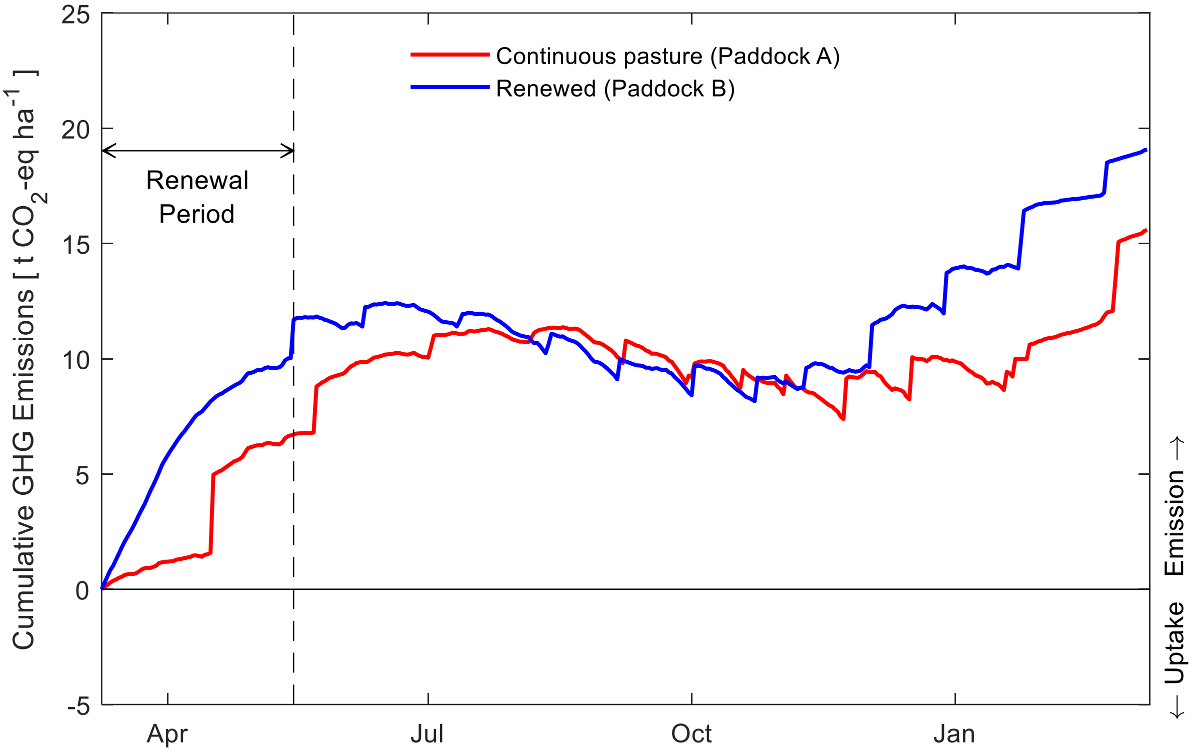

Figure 3: Cumulative GHG emissions (including the change in ecosystem C storage) from two adjacent paddocks with one undergoing pasture renewal. The renewal period extended from the time of herbicide application to the day before the first grazing after renewal. Vertical steps in the cumulative emission curves indicate grazing events.

During the remainder of the year following the pasture renewal period, the GHG emissions of the renewed pasture were influenced by management. Halfway through the year, the cumulative emissions of the two paddocks were similar (Figure 3), partially due to the increased CO_2 uptake of the new pasture sward relative to the established pasture, but also due to the conservative grazing regime implemented by the farmer to protect the new sward from damage. For example, the reduced grazing duration of the renewed pasture resulted in less excreta deposition, and thus lower N_2O emissions. Towards the end of the year, GHG emissions from the renewed paddock again increased relative to the continuous pasture paddock and were driven by lower net CO_2 uptake. We attributed the lower net CO_2 uptake to the poor establishment of the plantain included in the new pasture sward (see Figure 4a).

Effect of plantain on GHG emissions

The primary purpose of the pasture renewal was to establish a sward that contained greater than 30% plantain. Plantain has plant traits demonstrated to be beneficial for reducing N_2O emissions from grazed pastures (de Klein *et al.*, 2019), with a 30% proportion considered sufficient to have a measurable effect. Unfortunately, the poor establishment of the plantain in our trial resulted in difficulty achieving this targeted minimum plantain proportion (Figure 4a). Oversowing with additional plantain one year later, in autumn 2019, temporarily boosted the plantain to this 30% level, but this only persisted for around six months.

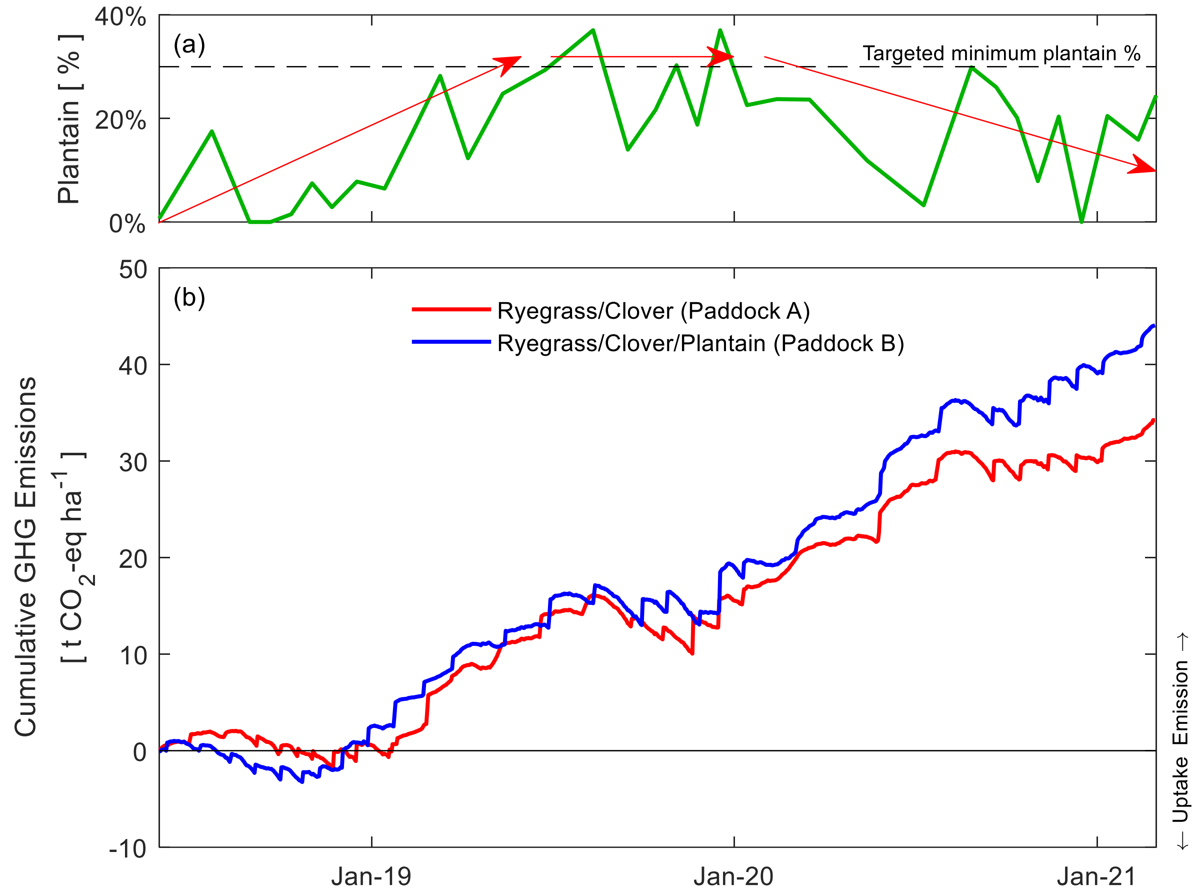

Figure 4: (a) proportion of plantain in the sward (as % of dry matter) in the ryegrass/clover/plantain sward. The red arrows indicate the general trends of plantain in the paddock; (b) cumulative GHG emissions (including the change in ecosystem C storage) from the ryegrass/clover and ryegrass/clover/plantain swards.

Although there were short periods of difference in the cumulative GHG emissions between the two pasture swards, there was no significant trend, and in autumn of 2020, two years after the establishment of the plantain sward, the accumulated GHG emissions were almost the same (Figure 4b). However, in the last year of measurements (from March 2020 onwards), the ryegrass/clover/plantain sward emitted considerably more GHG. This point of divergence in the cumulative GHG emissions coincided with decreasing plantain proportions in the sward. It should be reiterated that these data are still preliminary, and further analysis is pending.

Conclusions

Measured GHG emissions from two Waikato agricultural pastures ranged from 13.9 to 22.4 t CO_2-eq ha^{-1} y^{-1}. New Zealand's GHG inventory includes emissions attributed to the change in the ecosystem, or soil, C storage for managed organic soils, but not for mineral soils. At our site, the change in ecosystem C storage accounted for 50% and ~40% of the total GHG emissions from the organic and mineral soil sites respectively. However, analysis of available C balance data indicated that much larger C loss can be observed from organic soils, and mineral soils are likely at a steady-state (zero change in C). Consequently, our organic and mineral soil sites may, respectively, underestimate and overestimate long-term GHG emissions. Including ecosystem C changes in GHG emissions estimates are essential for organic soils and can influence annual emissions from minerals soils, but is likely to be negligible for long-term GHG emissions from mineral soils under continuous pasture. Management practices such as pasture renewal can increase GHG emissions, while longer-term (e.g. annual) GHG emissions are further influenced by day-to-day management such as grazing decisions. Finally, we established a pasture sward containing ryegrass, clover and a target plantain proportion of >30% to allow comparison of GHG emission with a typical ryegrass/clover sward, however, poor establishment and retention of the plantain make it difficult to draw conclusions from the preliminarily results at our site.

Acknowledgements

We acknowledge funding for this research from the New Zealand Greenhouse Gas Research Centre, the Ministry for Primary Industries, and the University of Waikato. We thank the farm owners, the Troughton family and JD and RD Wallace Ltd for access to their farms and management records. Technical and field support was provided by Chris Morcom, Anne Wecking, Liyin Liang, Jack Pronger, Georgie Glover-Clark, Seager Ray and Ben Roche.

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