Manaaki Whenua
Landcare Research

Determining the greenhouse gas reduction potential of Regenerative Agricultural practices

Prepared for:
Our Land and Water National Science Challenge & the NEXT Foundation

May 2021

'Think piece' on Regenerative Agriculture in Aotearoa New Zealand: project overview and statement of purpose

Gwen Grelet & Sam Lang

Extracted from: Grelet GA, Lang S 2021. 'Think piece' on Regenerative Agriculture in Aotearoa New Zealand: project overview and statement of purpose. Manaaki Whenua – Landcare Research Contract Report LC3954 for Our Land and Water National Science Challenge & The NEXT Foundation.

Find the full project overview, white paper and topic reports at

ourlandandwater.nz/regenag and www.landcareresearch.co.nz/publications/regenag

This report is one of a series of topic reports written as part of a 'think piece' project on Regenerative Agriculture (RA) in Aotearoa New Zealand (NZ). This think piece aims to provide a framework that can be used to develop a scientific evidence base and research questions specific to RA. It is the result of a large collaborative effort across the New Zealand agri-food system over the course of 6 months in 2020 that included representatives of the research community, farming industry bodies, farmers and RA practitioners, consultants, governmental organisations, and the social/environmental entrepreneurial sector.

The think piece outputs included this series of topic reports and a white paper providing a high-level summary of the context and main outcomes from each topic report. All topic reports have been peer-reviewed by at least one named topic expert and the relevant research portfolio leader within MWLR.

Foreword from the project leads

Regenerative Agriculture (RA) is emerging as a grassroot-led movement that extends far beyond the farmgate. Underpinning the movement is a vision of agriculture that regenerates the natural world while producing 'nutrient-dense' food and providing farmers with good livelihoods. There are a growing number of farmers, NGOs, governmental institutions, and big corporations backing RA as a solution to many of the systemic challenges faced by humanity, including climate change, food system disfunction, biodiversity loss and human health (to name a few). It has now become a movement. Momentum is building at all levels of the food supply and value chain. Now is an exciting time for scientists and practitioners to work together towards a better understanding of RA, and what benefits may or not arise from the adoption of RA in NZ.

RA's definitions are fluid and numerous – and vary depending on places and cultures. The lack of a crystal-clear definition makes it a challenging study subject. RA is not a 'thing' that can be put in a clearly defined experimental box nor be dissected methodically. In a way, RA calls for a more prominent acknowledgement of the diversity and creativity that is characteristic of farming – a call for reclaiming farming not only as a skilled profession but also as an art, constantly evolving and adapting, based on a multitude of theoretical and practical expertise.

RA research can similarly enact itself as a braided river of interlinked disciplines and knowledge types, spanning all aspects of health (planet, people, and economy) – where curiosity and open-mindedness prevail. The intent for this think piece was to explore and demonstrate what this braided river could look like in the context of a short-term (6 month) research project. It is with this intent that Sam Lang and Gwen Grelet have initially approached the many collaborators that contributed to this series of topic reports – for all bring their unique knowledge, expertise, values and worldviews or perspectives on the topic of RA.

How was the work stream of this think piece organised?

The project's structure was jointly designed by a project steering committee comprised of the two project leads (Dr Gwen Grelet[1] and Sam Lang[2]); a representative of the New Zealand Ministry for Primary Industries (Sustainable Food and Fibre Futures lead Jeremy Pos); OLW's Director (Dr Ken Taylor and then Dr Jenny Webster-Brown), chief scientist (Professor Rich McDowell), and Kaihāpai Māori (Naomi Aporo); NEXT's environmental director (Jan Hania); and MWLR's General Manager Science and knowledge translation (Graham Sevicke-Jones). OLW's science theme leader for the programme 'Incentives for change' (Dr Bill Kaye-Blake) oversaw the project from start to completion.

The work stream was modular and essentially inspired by theories underpinning agent-based modelling (Gilbert 2008) that have been developed to study coupled human and nature systems, by which the actions and interactions of multiple actors within a complex system are implicitly recognised as being autonomous, and characterised by unique traits (e.g. methodological approaches, world views, values, goals, etc.) while interacting with each other through prescribed rules (An 2012).

Multiple working groups were formed, each deliberately including a single type of actor (e.g. researchers and technical experts only or regenerative practitioners only) or as wide a variety of actors as possible (e.g. representatives of multiple professions within an agricultural sector). The groups were tasked with making specific contributions to the think piece. While the tasks performed by each group were prescribed by the project lead researchers, each group had a high level of autonomy in the manner it chose to assemble, operate, and deliver its contribution to the think piece. Typically, the groups deployed methods such as literature and website reviews, online focus groups, online workshops, thematic analyses, and iterative feedback between groups as time permitted (given the short duration of the project).

---

[1] Senior scientist at MWLR, with a background in soil ecology and plant ecophysiology - appointed as an un-paid member of Quorum Sense board of governors and part-time seconded to Toha Foundry while the think piece was being completed

[2] Sheep & beef farmer, independent social researcher, and project extension manager for Quorum Sense

Determining the greenhouse gas reduction potential of Regenerative Agricultural practices

Contract Report: LC3954-12

Johannes Laubach[1], Paul Mudge[1], Sam McNally[2], Pierre Roudier[1] and Gwen-Aëlle Grelet[1]

[1] *Manaaki Whenua – Landcare Research*

[2] *Plant & Food Research*

Please cite as follows: Laubach J, Mudge P, McNally S, Roudier P, Grelet GA. 2021. Determining the greenhouse gas reduction potential of Regenerative Agricultural practices. Manaaki Whenua – Landcare Research Contract Report LC3954-12 for Our Land and Water National Science Challenge & The NEXT Foundation.

Reviewed by:	*Approved for release by:*
Peter Millard General Manager Science Manaaki Whenua – Landcare Research	John Triantafilis Portfolio Leader – Managing Land & Water Manaaki Whenua – Landcare Research

Contents

1 Introduction

Regenerative Agriculture (RA) can potentially contribute to mitigating climate change via a reduction of greenhouse gas (GHG) emissions from agriculture and increases of soil carbon stocks. The global potential for such reductions is considerable (Smith et al. 2014; Project Drawdown 2020), but the feasibility of such reductions is disputed among scientists (e.g. Loisel et al. 2019; Amundson & Biardeau 2019).

This report aims to give an overview of the processes known, or suspected, to achieve GHG reductions, and to describe how such contributions can be verified and quantified. The GHGs considered are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Claims for RA's mitigation potential have been made for each of these gases. The report will begin with an overview of the main facts about agricultural GHG emissions and soil carbon storage. This is followed by a section explaining the different claims for how RA can mitigate GHG emissions and the knowledge context on which each claim is based. Four main sections then discuss the practical challenges to investigating these claims and the available methods to overcome the challenges. The final section summarises the knowledge gaps and recommends priorities for future research.

2 Overview of agricultural GHG emissions in Aotearoa New Zealand

New Zealand's GHG emissions are heavily influenced by agricultural activities, which account for 48% of total emissions (MfE 2020). Figure 1 provides an overview of the processes contributing to the greenhouse gas balance of an agricultural system. The processes shown inside the rectangular frame occur on the agricultural land itself, while those shown outside the rectangle occur elsewhere. The former processes are discussed here, as are the associated net emissions usually considered as deriving directly from agriculture. The latter processes are not discussed here, though they may be included where the wider impact of agricultural activities is assessed, such as in life-cycle analyses. Figure 1 is drawn for animal agriculture, which is dominant in large parts of New Zealand, but it could be easily adapted for arable or horticultural systems by removing the animal-related processes in the top-left quarter of the figure.

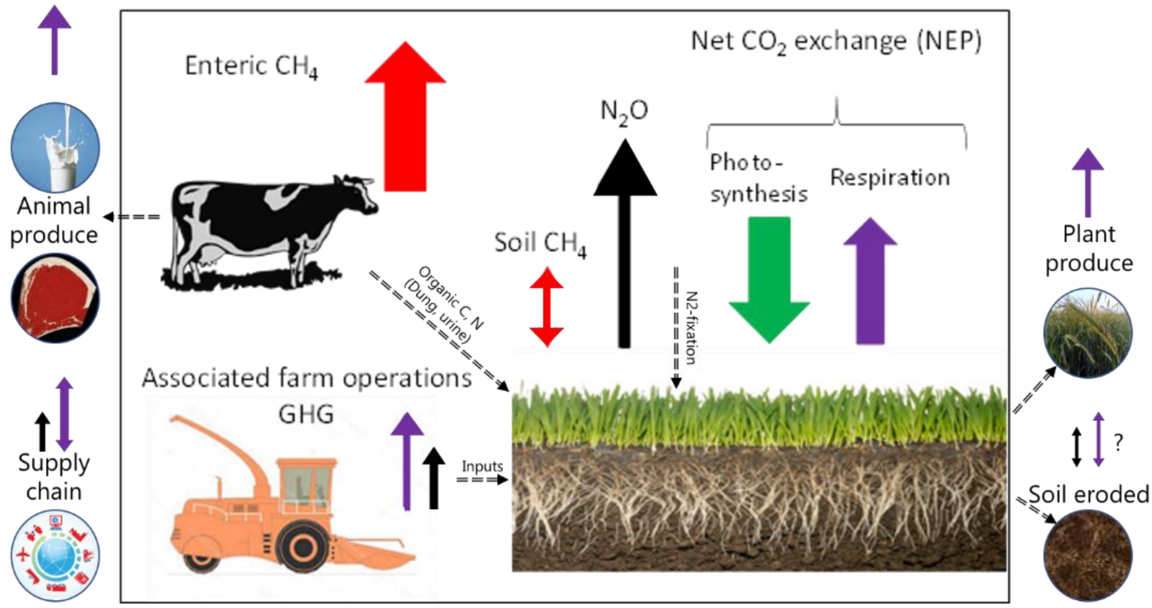

Figure 1. Schematic of the processes contributing to the greenhouse gas balance of an agricultural system with animals. Solid arrows indicate GHG flows, with their thickness giving a qualitative indication of the relative magnitude of the GHG exchange (in CO_2-equivalents). Dashed arrows indicate transfers of matter in other forms that provide source material for GHG emissions. Processes shown inside the rectangular frame occur on the agricultural land and are considered in this chapter. Processes outside would be accounted for in life-cycle assessments but are not considered here. For crop production, the animal-related processes shown in the top left do not occur, but the other parts of the schematic remain the same.

At the national scale, net GHG emissions from agriculture are dominated by CH_4 from animals, followed by N_2O emissions from soils, which are predominantly a consequence of the deposition of animal excreta and application of fertilisers (MfE 2020). CH_4 emissions from effluent and excreta deposition are generally small (two magnitudes smaller than enteric emissions). The same applies to uptake and emissions of CH_4 by soil organisms (indicated with the bidirectional arrow in Figure 1).

Net GHG emissions are also influenced by carbon sequestration or release from soils and vegetation. Increasing soil carbon stocks is proposed as one method to mitigate and reduce atmospheric CO_2 concentrations. To promote this, international initiatives (e.g. 4 per mille, Soils for Food Security and Climate) have been established, highlighting the significant role that soils can play in the mitigation of GHG emissions from agriculture. In general, soil carbon stocks are determined by the balance between carbon inputs to the soil (primarily from *in situ* photosynthetic uptake of atmospheric CO_2 by plant biomass), and losses through respiration of microbes and other soil biota (e.g. worms), and leaching of dissolved organic carbon. Increasing the stock of carbon relies on increasing the input of carbon to soil, or decreasing the losses of carbon from the soil, ideally with both occurring.

The stability of carbon retained in the soil can also be important if it is to have a long-term effect on atmospheric CO_2 concentrations. If carbon is not in a stable form, it can be more easily released later in response to changes in climate or management. New Zealand soils have high soil carbon stocks relative to many other countries. For example, on average soil carbon stocks in the top 30 cm of New Zealand soils are 90 t/ha, compared with Australia with 30 t/ha, the US with 70 t/ha, and France at 68 t/ha (FAO 2019a). However, it is important to note that from a GHG perspective it is changes in soil carbon stocks that are important rather than the total stock. There is good evidence that changes in land use in New Zealand increase or decrease soil carbon stocks (McNeill et al. 2014), but there is currently limited published evidence that specific management practices within a land use have a big impact on soil carbon stocks. This is an area of active research.

It is not only soil carbon stock that changes: the net GHG emissions from the other processes in Figure 1 can be altered by farm management practices. The various ways in which RA management may do so are discussed in the following section. A point worth noting is that when interpreting results for GHG emissions from comparison studies, it should be clear whether they are reported per area, per animal numbers, or per product output. Ideally, all three should be considered when making judgements on whether an observed effect is beneficial or not. Further steps would be to consider GHG emission per unit financial gain, and to include the benefits of GHG mitigation in economic assessments of RA management.

3 Claims for how RA could reduce GHGs

A number of different claims have been made how RA could reduce net GHG emissions. In this section we briefly explore these claims. They are related to the core RA principles to:

- minimise bare soil and keep living roots in the ground as much as possible
- minimise disturbance (e.g. tillage)
- harness diversity (plants, microbes, insects, birds, livestock)
- reduce external (synthetic) inputs
- manage livestock strategically (e.g. adaptive multi-paddock grazing).

3.1 Claim 1: RA soils are a greater sink for carbon than other agricultural soils

In general, the core principles RA practitioners seek to follow are beneficial for maintaining and potentially increasing soil carbon stocks. When soils are left bare, there is a net loss of carbon because carbon inputs from photosynthesis stop, while carbon outputs via soil respiration continue (Rutledge et al. 2014). Bare soils are also more prone to erosion, which reduces soil carbon stocks, although the overall net effect of erosion on GHGs may not be negative, due to sequestration of eroded carbon and re-accumulation of carbon in soil at eroded sites (Dymond 2010; Wang et al. 2017). Preventing erosion is still critically important for maintaining wider soil and ecosystem function (Donovan et al. 2021).

Global meta-analyses have shown that, compared to conventional tillage, no or minimum tillage generally leads to higher carbon in surface soils, but the effect on carbon stocks in the full soil profile is influenced by climate, soil texture, and management practices (Ogle et al. 2019; Li et al. 2020; Sun et al. 2020). Ogle et al. (2019) concluded that any potential benefit of no-till on soil carbon stocks should be viewed as secondary to other benefits (e.g. reduced erosion). Increased plant functional diversity has been shown to increase soil carbon stocks in ungrazed grasslands (established on former arable land) in North America and Europe (Fornara & Tilman 2008; Lange et al. 2015), and there is some evidence that increased plant diversity improved the net carbon balance of an intensively grazed pasture in New Zealand (Rutledge et al. 2017).

There is also evidence from New Zealand that long-term differences in fertiliser inputs influence root biomass and below-ground carbon allocation (Saggar et al. 1997; Scott et al. 2012), but this did not result in differences in soil carbon stocks (Lambert et al. 2000; Condron et al. 2012). Due to limited cropland area in New Zealand, the effect of altering grazing management in our grassland ecosystems is where any potential benefits will likely be greatest, at least in the near term.

Over the last 150 years or so New Zealand has championed the development of rotational and deferred grazing practices. Key differences between these leading grazing practices and grazing management strategies adopted by regenerative practitioners are that RA (i) focuses on having higher plant biomass both before and after grazing, and (ii) purposely manages both animals and vegetation to promote more diverse pastures with lower external inputs (including supplementary feed). Higher post-grazing biomass (lower utilisation) means that in the short term more fixed carbon is retained in the system (e.g. as litter), which could potentially become incorporated into soil organic matter in the longer term.

While there is some evidence that implementing individual RA principles could benefit soil carbon stocks, even in the international arena, there are limited published studies that have investigated a whole suite of practices applied together in a full farm system (which is in itself a core principle of RA). Early research on this topic, by Teague et al. (2011) in grazed US grasslands, reported an increase in soil organic matter (SOM) in systems implementing adaptive multi-paddock (AMP) grazing compared to other grazing regimes. In this study, SOM concentrations were measured throughout the soil profile to depths of up to 1.2 m, using the loss-on-ignition (LOI) method, and Wang et al. (2015) converted these values to SOC using a conversion factor of 0.58.

Jensen et al. (2018) point out that caution should be applied when interpreting SOC stock data calculated using the LOI method and standard conversion factor, which can grossly overestimate SOC stocks (although comparisons between treatments may be valid). Jensen et al. recommend high-temperature dry combustion methods. Bulk density (BD) was assessed in the top 10 cm of the soil profile and was not significantly affected by grazing practices. Conclusive evidence of soil carbon stocks requires an equivalent soil mass approach (Wendt & Hauser 2013). Hence, assuming BD also remains unchanged below 10 cm depth, the reported increase in SOM concentration could be interpreted as an increase in soil carbon stock under AMP grazing. However, some further verification might be needed due to the methodological limitations of this study.

A more recent study by Stanley et al. (2018) also reported increases in soil carbon stocks under AMP grazing, although there was no control treatment to compare results with. An ongoing, still unpublished US study, comparing AMP with mainstream grazing management across multiple paired farms, suggests SOC increases under AMP grazing (Teague 2020).

While there is currently no quantitative peer-reviewed and published evidence in New Zealand for increased soil carbon sequestration under RA, this could be tested with two approaches: (1) directly quantifying the impact of RA on changes in soil carbon stocks, and

(2) quantifying full carbon balances of RA systems (including all carbon inputs and outputs). These approaches are described in more detail in Section 5.

3.2 Claim 2: Increased plant diversity with better nutritional quality leads to a reduction of CH_4 emissions from animals

Regarding enteric CH_4 emissions from animals, it is a worthwhile hypothesis to investigate whether RA-based multi-species animal diet has the potential to reduce these, compared with the common ryegrass-clover dominated diet in New Zealand grazing systems. A US-based study with beef cattle indicated that the CH_4 emission factor (the mass of CH_4 emitted per mass of dry-matter intake) of a multi-species diet did not differ from that of a two-species alfalfa/grass diet, but the animals on the former diet had lower feed intake and therefore lower emissions per head (Thompson et al. 2019). In a New Zealand study comparing a group of dairy cows grazing fresh ryegrass-clover pasture with one grazing a five-species mix, while the latter group had smaller (but not significantly smaller) dry-matter intake, it also had greater CH_4 emissions per animal and per dry-matter intake. However, there was no difference in CH_4 emissions by milk production, because the diverse-feed cows were more productive (Jonker et al. 2018). The same authors also discuss emission factor results from studies in animal respiration chambers, which generally report little variation with plant species.

Some common practices of RA use fish- or seaweed-containing products, either as components of 'biological fertilisers' aimed at increasing plant health, or as feed additives for animals. The seaweed *Asparagopsis*, when added to the feed of dairy cows, has been found to very effectively reduce their CH_4 emissions (Roque et al. 2019), and the potential for producing effective seaweed extracts in New Zealand is already under investigation. As with other farm inputs, the net benefits of reducing methane emission by cows with feed additive must be assessed after taking into account GHG emission from harvest, extraction, and transport of source feed material. It is thus not impossible that other fish or seaweed products already on the market also have CH_4-mitigating effects, which would need to be tested individually for each such product. So far, however, only *Asparagopsis* has been demonstrated to remain effective and dramatically anti-methanogenic without negative impacts on rumen function and at low inclusion levels in animal diets (Roque et al. 2019).

Numerous tested feed supplement options of land-based or synthetic origin have been found to provide temporary mitigation effects, but after some period of adaptation rumen microbiology often reverts to original CH_4 emission levels. (Some compounds that promise longer-lasting effects are under investigation by the Pastoral Greenhouse Gas Research Consortium, but details are not disclosed to the public because they are commercially sensitive.) Selecting and utilising high-quality forages with more starch and less fibre is thus considered to be an immediate and sustainable option, because this not only reduces CH_4 emissions per feed intake but also forms a basis for higher feed intake and higher production per animal (Haque 2018).

In RA practice, the selection and utilisation of feed would largely be left to the animals when grazing diverse pasture, although supplementation could further contribute to high-quality feed composition. In a review of CH_4 emissions from grazing beef systems, Thompson and Rowntree (2020) concluded that 'Producer decisions affecting the soil–plant–animal interrelationships show promise in reducing the CH_4 emission rates from cattle', in particular where the farmer's strategy is the long-term improvement of forage quality. RA practices are compatible with that, but the studies cited earlier (Jonker et al. 2018; Thompson et al. 2019) suggest that progress to reduce CH_4 emissions may be small and incremental. Hence, more on-farm experiments in New Zealand conditions would be required to establish whether RA management can systematically lower feed intake, CH_4 emissions per feed intake, or CH_4 emissions per produce output, and which RA practices are most effective in this regard.

3.3 Claim 3: RA soils have high methanotrophic capability

Soils are habitats for both methanogenic (CH_4-producing) and methanotrophic (CH_4-consuming) microbes. The former are archaea that require strictly anaerobic conditions (Nazaries et al. 2013), and they are responsible for CH_4 emissions from waterlogged soils (e.g. in bogs and rice paddies). The latter comprise many different kinds of bacteria that require aerobic conditions (Nazaries et al. 2013). Both may be present in farmed soils, occupying different micro-habitats. Soil structure (density and porosity) and soil water content determine the distribution of aerobic and anaerobic compartments, and thus of potential micro-habitats. Consequently, they are important variables influencing the balance of methanogenic and methanotrophic processes, which in turn determines whether the soil ecosystem behaves as a source or sink. RA management may potentially reduce net CH_4 emissions, or increase net CH_4 uptake where it leads to increased soil aeration (e.g. by minimising soil compaction) or to reduced waterlogging (e.g. by reducing irrigation needs).

Mineral soils under grassland are often found to be small net CH_4 sinks (Schaufler et al. 2010; Nichols et al. 2016), including in New Zealand (Li & Kelliher 2007; Saggar et al. 2007). Application of animal urine to the soil locally increases soil water content, increasing methanogenic activity and reducing the soil's CH_4 uptake (Li & Kelliher 2007; Nichols et al. 2016). Urine also provides a nitrogen source, as do fertilisers, and the interplay with biological CH_4 processes is complex, with ammonium and nitrate having different effects (Nazaries et al. 2013). RA management tends to reduce nitrogen inputs overall, which could possibly work in favour of CH_4 uptake. However, more research is needed – both in controlled small-scale experiments and real-farm situations – to arrive at general conclusions.

There seems to be great hope in the RA community that their management approaches will increase CH_4 consumption by soils. A recent US study (Dowhower et al. 2020) showed that AMP grazing is associated with greater net soil CH_4 uptake than continuous-grazing management practices. However, these results are not directly transferable to the New Zealand context, where rotational grazing is a well-established practice of conventional farming and much more common than continuous grazing, at least on dairy farms. It is possible that the application of RA grazing strategies of promoting aeration of soils and greater standing biomass than in conventional rotational grazing, both before and after grazing, could increase CH_4 consumption by soils. However, this is untested in New Zealand.

In New Zealand, a major point of difference between RA and conventional farming is the plant diversity rather than the grazing management. Studies into the effects of plant diversity on CH_4 exchange of grazed pasture in New Zealand are lacking. Increased plant diversity under RA management may affect microbial species diversity and abundance, with possible effects on CH_4 production and consumption, but these effects do not necessarily mean increased net uptake. For example, Niklaus et al. (2016) found in a 2-year experiment that net CH_4 consumption decreased while the number of plant species increased from 1 to 16, which the authors explained by an increase of soil moisture along with plant diversity.

De Vries et al. (2013) report a decrease in CH_4 consumption with decreased fungal to bacterial ratio in a European study across four countries and three land uses (intensive and extensive maize crop rotations and grasslands). As RA management places strong emphasis on increasing the fungal to bacterial ratio, regardless of land use, associated changes in soil CH_4 consumption might occur, although the magnitude of the change is likely to remain small.

In grazed grasslands, the CH_4 uptake or emissions per area by the soil is of the order of a few kg CH_4/ha/yr (Smith et al. 2000; Li & Kelliher 2007), or even smaller (as in Dowhower et al. 2020). Improvements in net CH_4 uptake would need to be seen in the context of enteric CH_4 emissions from animals grazing the same land area. In New Zealand, typical stocking rates on a whole-farm basis are two to four cows per hectare, and a typical dairy cow in New Zealand emits about 100 kg CH_4/yr, which means the animal emissions per area exceed the CH_4 exchange of the soil by one to three magnitudes. That makes it challenging to detect changes of soil CH_4 uptake or emissions due to management, and it also limits the overall GHG mitigation potential of such changes. Reductions in stocking rate would probably have stronger benefits on the land's net CH_4 budget than increases in soil methanotrophy.

3.4 Claim 4: Regen soils produce less nitrous oxide

Nitrogen inputs to agricultural soils are predominantly from fertilisers, animal excreta, atmospheric nitrogen deposition, and microbes associated with nitrogen-fixing plants such as clover or lucerne. In the soil, various nitrogen compounds are transformed via chemical and biochemical reactions, most of which are part of the reaction chains of nitrification and denitrification (Butterbach-Bahl et al. 2013). In both these chains, as well as in a few less-common competing reaction chains, N_2O is a by-product of certain biochemical reaction steps, with different groups of bacteria as the main actors (van Groenigen et al. 2015), and fungi also involved in some competing reactions (Butterbach-Bahl et al. 2013). The amount of emitted N_2O is a fraction of the amount of nitrogen applied in fertiliser or deposited in animal excreta, typically in the order of 1%. Because of that, it is reasonable to expect that farm management approaches operating with reduced amounts of nitrogen fertiliser or reduced nitrogen deposition in animal excreta will lead to reduced N_2O emissions.

N_2O emissions are greatest at relatively high – but not fully saturated – soil water content (van der Weerden et al. 2012). The evidence is mounting that peak emissions occur at a specific, rather low value of relative soil gas diffusivity (Balaine et al. 2016; Rousset et al. 2020). High soil gas diffusivity is correlated with high soil air content (Rousset et al. 2020), at which level N_2O emissions are small. Farm management practices that lead to enhanced soil aeration are thus likely to also reduce N_2O emissions. However, in any soil that receives rainfall regularly there will be periods following rain events in which the relative soil gas diffusivity is close to its peak-emission value. How much N_2O emission then occurs will depend on how much nitrogen is available for denitrification at the time (for example, availability can be high soon after a grazing event). Because of this interplay of the timing of farming and weather events, research aimed at testing the claim that RA management reduces N_2O emissions would need to collect N_2O emission data over longer periods, covering all seasons.

There are further reasons why RA management could potentially alter N_2O emissions. One is the diversity of plant species. It has been found that the presence of plantain, *Plantago lanceolata* L., in mixed-species pasture can reduce N_2O emissions from cattle urine (Luo et al. 2018; Simon et al. 2019). Whether the observed reductions in N_2O emissions were due to diuretic effects on the animals leading to lower urine-nitrogen concentrations, reduced feed-nitrogen intake, direct inhibition of N_2O formation in the soil by the action of root exudates from the plantain, or some combination of these mechanisms is not fully understood (Simon et al. 2019; Pijlman et al. 2020).

It is possible that in a highly diverse species mix, other plants with biological nitrification-inhibiting potential may exist, or that some plants can take nitrogen up more quickly than others, which would reduce the availability for microbial processes that produce N_2O. Niklaus et al. (2016) found that N_2O emissions decreased with increasing number of plant species. We are not aware of any New Zealand studies in this context.

It appears there is potential for, but insufficient knowledge about, benefits of RA management with regard to N_2O emission reductions. Studies, both at small-plot scale to improve process understanding and at field or paddock scale to gather data on the net effects of RA management, are needed to increase our knowledge and test the claims.

4 Measurement methods: general challenges and options

It is very difficult to study the effects of individual farming practices in isolation, because often a number of practices will need to be combined to achieve the desired outcomes for production or the environment (e.g. the choice of pasture species affects the choices of fertiliser and water regimes, animal-feeding practices, etc.). It is therefore the set of practices, termed 'management' in the following, that distinguishes RA from other farming approaches, while the conditions it operates in, such as climate and soil types, are considered as given. Quantification of the effects of management must therefore rely on comparing specified management types that operate in the same (or similar) climate and soil conditions and measuring differences between these.

Here we consider the challenges for such comparisons to determine the effects of management on greenhouse gas emissions and soil carbon changes, as well as general options to overcome these challenges. Available measurement approaches will then be discussed in subsequent sections.

A general challenge in experimental research is finding the right balance between, on the one hand, controlling external factors that affect the outcome of experiments enough to find clear results (be these causal or statistical), and on the other hand, allowing enough of the real-world environmental conditions to be unaltered so that the results are meaningful in practice. Laboratory experiments are at one end of the scale, with maximum control but also the greatest disturbance of and remoteness from real-world conditions. Small-plot experiments still allow control of various factors while making it possible to operate with undisturbed soil and real weather conditions. Neither, however, are able to represent the entire combination of a set of farming practices, and small-plot experiments face multiple challenges associated with appropriate spatial representation of GHG sources and sinks.

Traditional agricultural research typically addresses these challenges by using randomised small-plot trials. In some cases, 'farmlet' experiments to represent full farm systems at smaller scale are established on randomised paddocks and compared (Beukes et al. 2017). These are usually expensive, because they require long-term commitment and the separate and consistent application of all aspects of the farm-management systems to be represented. In effect, the research costs include the farming operation's costs, with limited scope for efficiency or profits. A cheaper alternative is to set up experiments on commercial farms, which provides the best realisation of real-world conditions and allows for measurements at whole-field or -paddock scale, maximising spatial representativeness. However, these experiments provide the least control of actual farm management and environmental factors, and in order to compare different farm managements one needs to operate on more than one farm.

Climatic differences between farm locations can be minimised by choosing paired farm sites close to each other. However, due to the high small-scale variability of physical and chemical soil properties and differences in management history, large numbers of paired sites are usually required to detect differences between management regimes. For example, at least 15 paired sites (within a few 100 m of each other) would be required to detect a soil carbon difference of about 5 t/ha, should a difference of that magnitude be present.

In addition to soil variability, different areas on a farm have different functions, which poses questions of how to integrate these when comparing between farms for management effects. Even for one type of area (e.g. one paddock) it is often unclear how representative measurements at a fixed point can be, because actual usage of the area by grazing animals may vary due to factors such as locations of gates, water sources, shade, etc. Also, GHG emissions are subject to strong, small-scale variations, not only because of the variability of soil properties, but also because the deposition of animal excreta during grazing is patchy: urine- or dung-covered areas behave very differently from unaffected soil areas, so when trying to quantify the gas exchange of a paddock, the coverage with excreta needs to be weighted appropriately.

There are two general options for dealing with the various challenges of spatial representation: multiple replication of small-scale measurements, or integration over larger areas by methods that can operate at the field/paddock or farm scale. Examples of both approaches are discussed further below.

Temporal coverage is another challenge. Management events that have big impacts on soil processes and their associated GHG emissions, such as sowing, grazing, harvest, or fertiliser application, can generally not be synchronised between farms, or even between paddocks on the same farm. Identifying differences between differently managed farm locations will therefore often require integration over whole farming years to become meaningful. Even one farming year can be too short, though, because effects of the timing of management events can be confounded by the timing of weather patterns. Also, the emission response to management events or precipitation events can be relatively rapid (confined to a few hours or days) and may not be adequately captured if sampling is not continuous. At the other extreme, total soil carbon stocks change relatively slowly, and generally require multiple years (at least three) before significant changes can be detected.

For processes that are not related to soil or weather, such as diet effects on animal CH_4 emissions, it may be possible to obtain meaningful comparisons over shorter periods and independent of location. The challenge here is how to ensure comparability between animals in differently managed farm systems. This requires the design of experiments with well-matched groups of animals and well-specified protocols for their feeding and other activities.

A general challenge with the comparison of farm management effects is that always only a limited number of specific management choices can be studied directly. Due to the multi-factorial nature of 'management', the interpolation between or extrapolation outside the range of studied cases is often not straightforward. Process-based models can be an important tool to overcome this challenge. Once such models are well calibrated and validated with a number of experimentally studied management options, they may give good guidance on the interdependence of soil carbon changes and GHG emissions with other outcomes, such as product yield, as demonstrated by Kirschbaum et al. (2017) for dairy farming.

5 Options to quantify changes in soil carbon stocks

Currently, two broad approaches are available to determine whether RA soils are a greater sink for carbon than other agricultural soils (Claim 1). The first is to quantify soil carbon stocks directly at different times, either from collected samples (soil cores) or with emerging remote-sensing methods. The second is to quantify all carbon inputs and outputs of the plant–soil system and interpret the balance of these as changes of the amounts of carbon stored in the soil. This approach is known as the carbon-balance approach (also 'full carbon balance' or 'net ecosystem carbon balance'). Details of the two approaches are described in the following two subsections.

5.1 Soil sampling

Direct soil sampling to determine how management practices influence changes in soil carbon stocks can be achieved using two different study designs. The first and most accurate method is to directly monitor changes in soil carbon stocks by repeat soil sampling and analysis through time. This can be applied at plot scale (Schipper et al. 2013), farm scale (Mudge et al. 2020) or national scale (Mudge 2019). Sampling sites need to be a representative, unbiased subset of the area of interest, and this is generally best achieved using random site selection.

For changes to be attributed to management practices, baseline soil carbon stocks need to be determined prior to changes in management (i.e. before transition to RA). However, even if a baseline is obtained, if the whole study area is transitioned to the new management regime, changes in soil carbon stocks may still not be able to be attributed to management, because they could be due to other factors such as changes in climate. Having a control treatment where management has not changed helps isolate the effects of management. It is critical that sampling sites across different treatments be balanced by factors that are known to affect soil carbon distribution, such as soil type, topography, and previous management history. This is especially important when sampling on commercial farms (more than in statistically designed small plot experiments).

A major limitation of directly monitoring changes in soil carbon with time is that it will generally take multiple (3–10) years before significant changes can be detected (if changes are indeed occurring). An alternative is to use a paired site or chronosequence approach, whereby space is substituted for time (Mudge et al. 2017; Sparling et al. 2014). With this approach, soil carbon stocks are quantified at adjacent sites (generally on commercial farms) that have been under different land uses or management regimes for multiple years (usually more than 5 years). The average difference in carbon stocks between the different treatments is assumed to be due to differences in management, and rates of divergence between treatments can be calculated if the time since management changed is known.

This approach enables an estimate of treatment effects to be obtained from a single sampling campaign rather than sampling and waiting years before resampling and obtaining results. However, a key assumption of this approach is that soil carbon stocks were the same under both treatments prior to changes in management, an assumption that cannot be verified. It can only be made likely to hold with careful site selection, to ensure soils and topography are well matched. With that, and with replication across multiple farms, this has proven to be an effective method (Jackman 1964; Schipper et al. 2013; Barnett et al. 2014; Sparling et al. 2014; Ward et al. 2016; Mudge et al. 2017). After the initial sampling, repeat monitoring at the same sites can still be employed and will enable direct quantification of the rates of carbon stock change under the different treatments.

Specific methods for soil sampling, processing, and analysis to quantify soil carbon stocks and stock changes have been extensively documented in a number of recent protocols (Australian Government 2018; FAO 2019b, 2020; Mudge et al. 2020) and are only very briefly mentioned here. The generally accepted standard method for quantifying soil carbon concentration is high temperature combustion (of the <2 mm soil fraction) with an elemental analyser (FAO 2019b). The carbon concentration must be corrected to an oven-dry basis and carbon stocks calculated using soil bulk density. In addition to soil carbon stocks, a range of methods are available to determine how different management regimes influence the type and way carbon is stored in soils (Lavallee et al. 2020). How carbon is stored in soils has implications for long-term stability in the soil and its influence on other soil functions (e.g. nutrient cycling).

Among the carbon measurement techniques, soil spectroscopy has recently been under the limelight (Viscarra Rossel & Bouma 2016; Smith et al. 2020). Carbon, along with other properties, is estimated based on the reflectance of the soil surface in many different wavelengths (Stenberg et al. 2010). Two amin technologies are currently in use: near infra-red spectroscopy (NIR) and mid-infrared spectroscopy (MIR), which have demonstrated capability to generate estimates of soil carbon in a cost- and time-efficient fashion (Baldock et al. 2018).

The quality of the predictions depends largely on the quality and relevance of a calibration set used to parameterise a statistical carbon model. To determine changes in soil carbon stocks with time, it is critical that the same calibration set be used for all samples. These proximal sensing techniques are not there to replace standard measurements as much as to allow for more samples to be tested, in order to better capture the carbon variations in space and/or time (Smith et al. 2020).

Remote sensing has been proposed more recently as a means to provide SOC estimates across large swathes of land. Archives of satellite imagery can be used as a predictor for spatial SOC models, to monitor vegetative activity and associated changes in SOC, or, more recently, to directly relate SOC content with the 'bare earth' reflectance (i.e. the colour of the least-vegetated pixel calculated across multiple years of data) (Gholizadeh et al. 2018; Chabrillat et al. 2019). Further advances will be required before accurate quantification of full-profile soil carbon stocks and stock changes can be determined from remote sensing (particularly when soils are always vegetated, which is a key principle of RA).

5.2 Carbon balances

The second approach is to quantify the full field- or paddock-scale carbon balance of the different farm systems by determining all carbon inputs to and outputs from the plant–soil system (Figure 2). The inputs include CO_2 uptake by photosynthesis and, potentially, the carbon contained in applications of lime and fertilisers. Major outputs are CO_2 respired by plants and soil organisms, and can also be the biomass removed by harvest; minor outputs can occur via leaching of dissolved carbon. Methane emissions and uptake by the soil constitute carbon outputs and inputs, respectively, but their relative contributions on a carbon basis are negligible compared to CO_2 inputs and outputs by photosynthesis and respiration.

The net exchange of CO_2 is, most commonly, measured continuously with a micrometeorological technique known as the eddy-covariance method, using a combination of a fast-responding gas analyser (for CO_2 concentration) with a three-dimensional ultrasonic anemometer (for wind speed in all directions, including the vertical). The other inputs and outputs, in solid or liquid form, need to be determined as appropriate (often by sampling before and after grazing events or other management actions, or from farm records of such actions).

For grazed paddocks, the role of the animals can be accounted for in two ways. If the animals are considered outside the plant–soil system (Hunt et al. 2016; Laubach et al. 2019), then they are agents of import, by depositing excreta, and of export, by removing biomass, and these terms need to be measured or estimated (as well as carbon in effluent recycled to the paddock). If the animals are considered as part of the system, as shown in Figure 2, then only conversion products of the digested biomass constitute exports (i.e. the produce, such as milk, meat, wool) and the CO_2 and CH_4 emitted by the animals during grazing (Rutledge et al. 2014, 2017), and supplementary feed needs to be counted as an additional input.

Neglecting supplementary feed, Figure 2 gives approximate annual figures for the carbon balance terms for a typical dairy farm paddock assumed to be about carbon-neutral. The two opposing flows of CO_2, associated with photosynthesis and respiration, exceed all other terms in magnitude. However, their difference, the net exchange of CO_2 of a field or paddock, can be quite small and of similar magnitude to the other inputs and outputs. Note that the arrows in Figure 2 show flows of carbon rather than CO_2-equivalents, as depicted in Figure 1, which is why the CH_4 arrow is so much smaller in Figure 2 than in Figure 1.

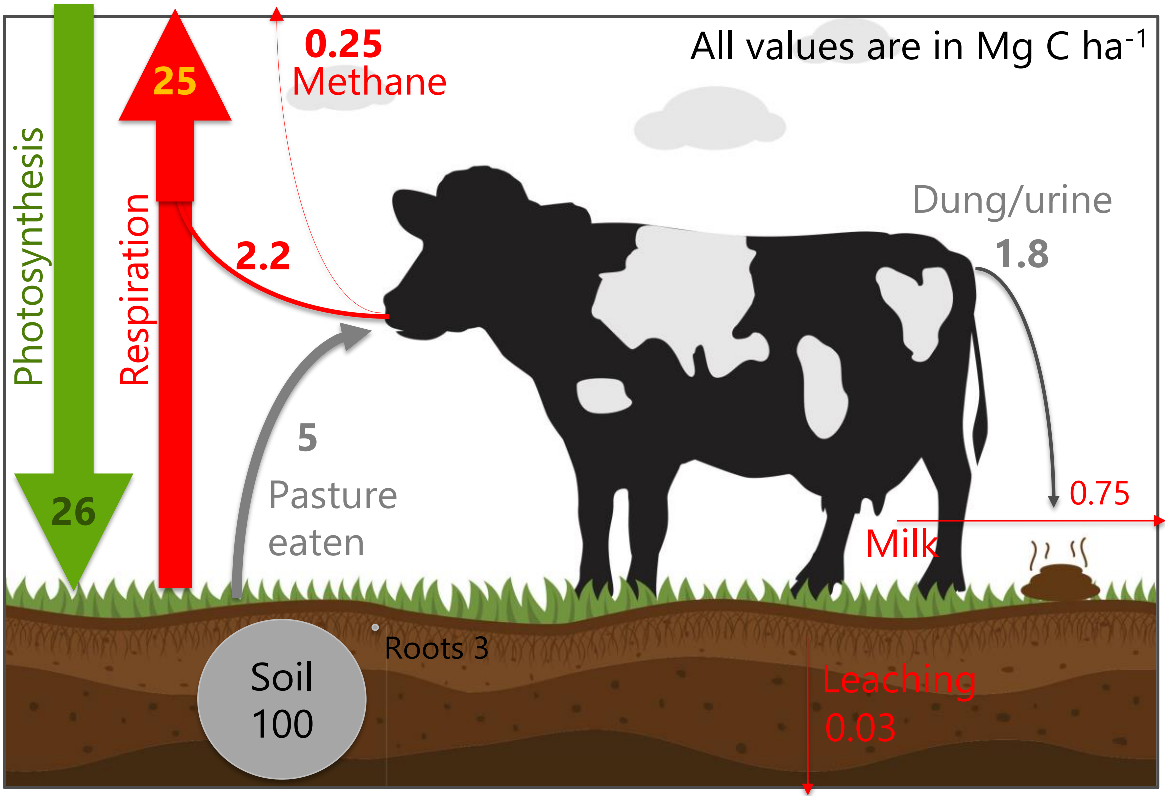

Figure 2. Key carbon inputs (green arrows), outputs and losses (red arrows) and internal cycling (grey arrows) in a typical New Zealand dairy farm paddock, for 1 year. Indicative carbon stocks in the top 0.3 m of soil are also shown. Values have been updated and simplified from Wall et al. 2020 (A. Wall, pers. comm., 26 Nov. 2020) and made to represent a system at about steady state. For carbon to increase in the system, inputs must increase, outputs decrease, or both. The system shown has no imported or exported supplementary feed. (Note the "Methane" arrow in this figure is much narrower than in Figure 1 because it represents the carbon content of CH_4, not its CO_2-equivalent GHG effect, as in Figure 1).

Two major advantages of the paddock-scale carbon balance approach are that (i) the carbon input and output data are obtained with high temporal resolution (sub-hourly for CO_2, event-based for the others), which helps elucidate specific causes for any observed differences between measurement sites, and thus to identify effects of management; and (ii) measurements are integrated over spatial scales relevant to management (hectares). The eddy-covariance method is well established around the world and standardised to a considerable degree, with hundreds of active measurement stations in the FLUXNET network,[3] of which the majority are in forests and natural ecosystems and a minority at agricultural sites.

[3] https://fluxnet.org/sites/site-summary/

Because of the need to install and maintain dedicated instrumentation at each site for whole years, both the capital and operating costs associated with eddy covariance can limit replication. On the other hand, an advantage of the approach is that over the last few years new instruments have become available that can simultaneously measure not only the CO_2 and H_2O exchange, but also N_2O and CH_4. This is invaluable when exploring how management influences trade-offs between different GHGs, and also for the calibration and validation of models and remote-sensing techniques; however, due to the costs of these specialised instruments and their power requirements, this will only be feasible at very few sites.

Another variant of the carbon balance approach, without eddy covariance, can be realised using chambers to measure CO_2 inputs and outputs. Both manual and automated chamber systems are available (their general advantages and disadvantages are discussed in the next section). For net CO_2 exchange, specifically, transparent and opaque chambers need to be combined. While manual chamber systems can have lower capital outlay than eddy covariance and automated chambers, they are likely to have high operating costs because of the need for frequent repetition of sampling campaigns throughout all seasons and in various weather conditions, and the scaling up from the campaign observations to whole-year balances has considerable uncertainty. Automated chambers can operate continuously, reducing this uncertainty, but have power and maintenance requirements comparable to eddy covariance, and the measurements represent only a small area. Chamber instrumentation must be protected from harm by farm machinery or animals, which poses additional challenges for reproducing management events, particularly of biomass removal, inside the chambers in a representative fashion.

A specific difficulty for using CO_2 chambers in RA systems can be the presence of rather tall vegetation. This is a challenge because CO_2 uptake is controlled by absorption of incoming radiation and can only be measured in representative fashion if the plants' positions and orientations are the same as they would be without the presence of the chamber. In other words, the plants must not be bent or compressed to make them fit into the chamber. (This is not a problem for N_2O and CH_4 exchange measurements [see next section], for which the sources and sinks are located in the soil.)

6 Options to quantify N_2O and CH_4 exchange of crops and pastures

6.1 Chamber methods

The most widespread methods to assess GHG emissions from soils, including crop and pasture soils, are chamber methods. Chamber frames are inserted into the ground prior to the experimental period. Measurements are only carried out at certain specified points in time. Then, each chamber is closed with a lid, for a period in which surface emission of the gas of interest will increase the gas's concentration in the chamber (or surface uptake will reduce it). Air samples are taken a few times during the closure period and analysed in the laboratory for the gas concentration differences over time, which give the gas exchange rate. Comprehensive international guidelines for all aspects of this method have been developed (see de Klein et al. 2020, and further articles in same Special Issue).

Chamber methods are best suited to controlled experiments, in which treatments in each chamber are specified (e.g. application of a known amount of urine with known composition) and the GHG emissions measured in response to the treatments. They are less well suited to comparisons of different 'managements', which involve a multitude of interdependent treatment differences. Most commonly, the sample collection is done manually, which means low investment costs and easy installation, but relatively high costs for labour and chemical analyses. The latter restrict in practice the numbers of replicates and the sampling frequency; consequently, manual chambers are not well suited to deal with the challenges of spatial representation and temporal coverage on real paddocks or fields.

Automated chamber systems are an alternative, using a dedicated gas analyser on-site, to obtain continuous coverage in time, but for investment cost and operational reasons the number of chambers in a system is limited, typically to small multiples of four (Grace et al. 2020), and the chambers need to be connected by piping, which severely restricts the size of the sampling area. A further challenge for chamber methods, specific to RA, is that sometimes rather tall plant species, such as sunflowers, are included in the grazing area; these would be difficult to accommodate within chamber volumes.

6.2 Micrometeorological methods

There is a group of measurement methods for gas exchange that are based on knowledge of how gas emitted from a source location (or taken up by a sink process) is transported in the air near the source/sink location. Such methods are called 'micrometeorological methods'. Their merits and considerations for their optimisation have been reviewed by various authors (Denmead 2008; Harper et al. 2011; Hensen et al. 2013). They have in common that they combine measurements of gas concentrations in the air above or around a source/sink area with measurements of airflow properties (in the simplest case, just wind speed) to derive gas exchange rates.

This group of methods includes, among others, the eddy-covariance method, where gas concentration and vertical wind speed in one location are measured a few times per second to calculate turbulent transport rates of the gas directly; and the flux-gradient method, where gas concentration differences between two heights ('gradients') are measured and combined with a turbulent diffusion coefficient derived from suitable air flow measurements at or between the two heights. These methods can operate continuously, year-round, and the measured gas exchange rates are representative for an area of a few tens to a few hundreds of metres upwind from the instrument mast (the 'footprint', varying with wind direction). As a result, these methods are well suited to the field- or paddock-scale, and they inherently integrate over small-scale variability of soil properties and excreta deposition.

However, they do require an expensive gas analyser plus meteorological instruments to be installed at each measurement site, and (to date) most types of suitable gas analysers for CH_4 and N_2O involve the operation of pumps, lasers, and temperature stabilisation, requiring mains power. With the flux-gradient method, the need for one analyser per site can be stretched to one for two neighbouring paired sites, as successfully operated on a New Zealand dairy farm by Laubach and Hunt (2018), and potentially even to four adjoining areas (McMillan et al. 2014).

With the eddy-covariance method, a paired site setup ('split-footprint' approach) has been successfully tested for CO_2 by Wall et al. (2020) and is now in use for CH_4 and N_2O at two sites in New Zealand, operated by University of Waikato and by Manaaki Whenua – Landcare Research, in order to compare net GHG budgets for mixed-species pasture and ryegrass-clover pasture. Due to the high instrument costs and the fairly complex data analysis, it does not appear feasible to achieve greater replication of sites, which limits the potential for testing effects of different farm management practices on GHG emissions at a wider range of locations. It should also be noted that micrometeorological methods require flat terrain without major flow obstacles, such as buildings or tree rows.

7 Options to quantify enteric CH_4 emissions

Enteric CH_4 emissions do not depend on soil processes and weather influences, but on the amount and composition of the animal diet and animal physiology. Thus, it does not seem necessary to monitor these emissions over full years or whole farms, as long as it is possible to record feed intake and its composition so that the relationship of these variables with the emissions can be specified.

The measurement of emissions from animals can be attempted right at the source (individual animals), by using enclosed spaces around the animals, or by detecting emission plumes downwind of groups of animals. The following subsections follow this order to describe some options.

Measurements with individual animals are usually labour-intensive in order to get sufficient replication for statistically significant results, and care needs to be taken that the animals' behaviour is not altered too much from the natural behaviour the measurements are aimed to represent. By contrast, measurements of emission plumes can be done with micrometeorological methods, which can be automated to a high degree and do not affect the animals' behaviour. However, they are laborious to apply to New Zealand's livestock farming systems, because mast-based instrumentation would need to be moved in step with rotational grazing that changes paddock every day. Also, the movement of animals as point sources within a paddock area creates additional uncertainty about the emissions footprint, which increases the uncertainty of the derived emission rates.

Attempts to address this have been made (e.g. Felber et al. 2015) by combining the eddy-covariance method with geolocation (GPS) sensors on cows, but even with the accuracy gains from this, emission differences between animal groups subject to different management practices would be hard to establish statistically. Rather than the micrometeorological methods mentioned in earlier sections, we describe a different one, specifically optimised for comparing two groups of animals, in Section 7.4.

7.1 Sulphur-hexafluoride tracer method

The sulphur-hexafluoride (SF_6) tracer method (Johnson et al. 1994) has long been considered the reference method for freely grazing animals, because it does not require the animals to be in a particular location and places no restrictions on their feeding behaviour.

Prior to a measurement campaign, groups of selected animals need to be implanted with a capsule that slowly releases the gas SF_6. During the campaign, the animals wear U-shaped containers around their neck with a mechanism that collects a fraction of the air exhaled by the animal into this container. The containers are exchanged regularly (typically daily, e.g. at the milking shed) and the collected air is analysed for the concentrations of CH_4 and SF_6. Both gases' concentrations are proportional to their respective release rates inside the animal, and because the release rate of SF_6 is known, that of CH_4 can be derived.

The method is labour-intensive and requires skilled staff for animal handling, container preparation and laboratory analyses. Also, SF_6 is expensive and is itself a potent greenhouse gas, and this type of experiment requires animal ethics approval. Despite these factors limiting its widespread use, the method was employed many times across New Zealand (Lassey 2007; Pinares-Patiño et al. 2012, 2016), but its usage has declined in favour of the following two methods, which allow better control and measurement of the feed intake associated with the CH_4 emissions.

7.2 Animal respiration chambers

Today, AgResearch uses animal respiration chambers as the method of choice for experiments to investigate relationships between feed intake and CH_4 emissions (Muetzel & Clark 2015; Jonker et al. 2016). The chambers are purpose-built, in different sizes for different livestock categories. Each selected animal spends the whole measurement campaign in its own chamber. Feed intake is fully controlled and recorded, and CH_4 emissions are measured continuously while the chamber air is exchanged in a controlled fashion. Because it is a controlled setup, the accuracy of the results is unrivalled. Disadvantages are the operating costs for the chambers and skilled staff, the relatively small animal numbers in each trial, and, in the context of RA probably most importantly, the animals cannot behave as they would normally on a farm. Animal respiration chambers can thus be considered useful for establishing relationships between the composition of RA animal diet and CH_4 emissions, but application of the results to estimate enteric emissions from a farm would still be subject to the uncertainties of possibly different animal behaviour and not well-known feed amounts and composition.

7.3 GreenFeed

GreenFeed (C-lock Inc., Rapid City SD, USA) is an instrument developed for the purpose of taking frequent CH_4 and CO_2 concentration samples in air exhaled by animals. The animals are enticed by offered feed supplements to enter a half-enclosed space, attached to a free-standing device or a trailer, one animal at a time. The gas concentrations in this space are continuously monitored, as is the frequency and duration of animal visits and their intake of the feed offered. The instrument has been tested for grazing dairy cows in New Zealand (Waghorn et al. 2016). Like micrometeorological methods, it can be used to integrate over emissions from a group of animals while these pursue their normal grazing routine; however, also like these methods, it is an expensive piece of equipment that cannot easily be replicated, and there is an additional random element in the visiting times by individual animals, not unlike animal movements within the footprint of a micrometeorological setup.

7.4 Atmospheric dispersion model

A micrometeorological method to compare emissions from two groups of animals, over a few days at a time, was developed by Laubach et al. (2014). It combines CH_4 concentration measurements upwind and downwind of the areas occupied by the animals (which can be grazing strips or feed pads), with wind and turbulence measurements nearby. These observations are then used for statistical simulations of the airflow, computed with an atmospheric dispersion model (Flesch et al. 2004), to calculate where air parcels arriving at the locations of the concentration measurements have been in contact with the ground. With this information, the dispersion model converts the concentration measurements into emission rates inside the specified source area (i.e. the fenced paddock).

Laubach et al. (2014) showed that, with careful optimisation of the measurement geometry, using concentration measurements integrated along fence lines (rather than at a few points only), it is possible to resolve relative differences in emissions of 10% or greater between two groups of animals. The method does not affect the animals' normal grazing behaviour. It requires a single CH_4 analyser, combined with an elaborate (but not expensive) intake system, and an ultrasonic anemometer to provide wind speed, direction, and other airflow properties. The labour effort is mainly in setting up and moving equipment between grazing days, while data acquisition is automated and data analysis aided by the freely available dispersion-model software WindTrax.[4]

The atmospheric dispersion-model method could be applied for direct comparison of two groups of animals, one under RA management and the other under conventional management, while the two groups are grazing paddocks not too distant from each other. Where close proximity of paddocks or simultaneous grazing cannot be arranged, the method can still be used sequentially, following one group of animals for a number of grazing days, then following the other group (possibly on a different farm). As with all other methods to measure enteric CH_4 emissions, data on feed intake and composition need to be obtained for each animal group to provide context for interpreting the emission results.

8 Recommendations for research priorities

For all four claims discussed in Section 3, there is some evidence in the international literature that RA management could have positive effects to reduce GHGs in the atmosphere, compared to more conventional agricultural practices. Globally, the greatest potential appears to be the increasing of carbon stocks in soils that have previously been depleted (Project Drawdown 2020, see chapters 'Managed grazing' and 'Regenerative annual cropping'). It should be noted that on previously degraded soils, conventional 'best practice' will likely also lead to carbon stock increases, and it is *a priori* unknown to what extent RA practices would exceed these.

[4] http://www.thunderbeachscientific.com/.

There is also little doubt that reductions in fertiliser use lead to reductions in N_2O emissions, with the possible exception of grazed systems (RA or otherwise) that contain a large fraction of nitrogen-fixing legumes (in these, total nitrogen inputs to the system and thus urinary nitrogen deposition could be greater than in a conventional system, so N_2O could be greater, too). Other potential reduction effects for CH_4 emissions from animals because of changes to animal diet, and for all GHGs due to changes in the soils' structure, chemistry, and biology, appear more subtle and dependent on specific circumstances.

In New Zealand, quantitative evidence for the effects of RA management on net GHG emissions is lacking, for all the discussed claims. Obtaining quantitative evidence is possible using existing and emerging methods as presented in Sections 4 to 7. This will require careful study design, to ensure comparison experiments are meaningful (representing realistic farm management practices), are not confounded by soil and weather variability, and will yield statistically significant results.

We suggest the following research priorities to test the claims put forward in Section 3. As these priorities address different claims (or aspects of claims), they should ideally all be implemented. They are ranked here roughly in order of increasing sophistication required.

1. *Direct quantification of soil carbon stocks (Claim 1)*

Option A: Paired site study on existing RA vs 'conventional' systems across New Zealand. On multiple commercial farms (20 or more), sample full-profile soil carbon stocks at carefully matched paired sites, comparing RA and an adjacent, more conventionally managed system. This should give, relatively cheaply and rapidly (e.g. within about 1 year), a good indication of the potential of RA in New Zealand to sequester additional carbon. This is a well-proven research approach, based on the assumption that paired sites had equal carbon stocks at the time of the transition (i.e. substituting space for time). This may not be true at all sites, but the larger the number of paired sites sampled, the better this assumption is likely to hold on average. Soil carbon stocks can then continue to be monitored over time to determine if the trajectories of carbon stock changes differ under the different management regimes. Other soil and plant parameters could be measured at the same sites to link GHG to production outcomes and soil health.

Option B: Replicated, farmer-led/-managed, paddock-to-farm scale, designed paired site experiments across New Zealand. Multiple farms (20 or more) apply RA principles to one part of the farm (e.g. a paddock), while an adjacent area is managed more conventionally. Baseline soil carbon stocks (and other properties) are quantified prior to treatments being applied and then monitored through time. This is a similar study design to Option A, with the key differences being: (1) the treatments are purposefully chosen and randomly allocated at the outset, which reduces the chance of bias in initial soil properties between treatments, but (2) results will only become available after monitoring through time.

A major advantage of the study design in both options is that implementation is relatively cheap across sites, with a range of soil types and climatic conditions. The results are, therefore, more generalisable than detailed studies at one or a few sites.

Such studies can also identify where responses to management are likely to be greatest, and then more detailed (and expensive) research conducted at these sites (e.g. using approaches as outlined below). Study site locations could be further chosen to purposely include some of the lowest and highest starting soil carbon stocks, to assess whether the RA impact of soil carbon sequestration is linked to inherent soil carbon content. The choice of which to implement, or whether to implement a mixture of both, would need to be weighed up in a more detailed design phase; for example, (a) is reliant on enough suitable paired sites where RA has been implemented, and (b) on enough farmers willing to change part of their farm to RA.

It would also be possible to specifically incorporate sampling of additional sites under RA into the existing National Soil Carbon Monitoring System for Agricultural Land as a separate land use class (Mudge 2019).

2. *Comparative soil chamber experiments for N_2O and CH_4 exchange (Claims 3 and 4)*

At a few sites (which can be a subset of those used in priorities [a] and [b]), undertake in situ chamber studies comparing N2O emissions from RA and non-RA soils, with and without application of animal urine. Soil CH4 exchange measurements can be included in these at relatively little effort. Such paired studies should give a reasonably rapid indication of whether RA has the potential to alter net GHG emissions from soils (Claims 3 and 4). The more plant and soil properties that are measured at these sites simultaneously, the greater the potential to not only obtain statistical data on the N2O (and CH4) flows, but also to gain a better understanding of what extent they are influenced by plant diversity and altered soil properties.

3. *Paired paddock-scale studies for CO_2, N_2O and CH_4 exchange (Claims 1, 3 and 4)*

Because small-scale chamber studies are likely to investigate only a subset of the properties altered by RA, and are not continuous in time, they should be complemented by paddock-scale studies at a few paired farm sites. For carbon, this should be done with the eddy-covariance-based carbon balance approach. For N2O and soil CH4, it is hoped that emerging measurement methods (low-cost flux-gradient) will make such studies easier to replicate in order to include a variety of soil and climate conditions.

4. *On-farm experiments for CH_4 from animals (Claim 2)*

Undertake on-farm experiments to compare CH4 emissions from animals on diverse RA diet and ryegrass-clover diet, combined with accurate estimations of animal feed intake. These experiments will have to be carefully designed to ensure the differences can be detected and Claim 2 validated. Either the atmospheric dispersion model or the GreenFeed instrumentation can be used. Such studies could be combined with assessments of the nitrogen return in excreta, for both animal groups, which would provide an indication whether N2O emissions are likely to be reduced by RA diet (a potential co-benefit) or increased (a potential trade-off).

The ultimate 'gold standard' for assessing the effects of RA would be to combine as many of the previously listed paired approaches as possible in randomised and replicated trials with full farm systems, such as by Beukes et al. (2017), and monitor a whole suite of additional parameters (including production and profit) over multiple years. While this is technically possible, it would be expensive and require a fairly large consortium of researchers from various disciplines to coordinate their efforts. Clearly this could only be done at a very small number of sites, limiting the generalisation of results for the whole country and for all branches of the agricultural and horticultural sectors. We thus recommend that, at least initially, limited research funds are probably better applied to one or other of the above-listed priorities than to an all-inclusive full-farm system project.

Detailed small-plot experiments could be employed to disentangle specific individual mechanisms (e.g. differences in plant diversity or grazing management) from any observed differences between different systems studied in the priorities above.

9 References

Amundson R, Biardeau L 2019. Reply to Loisel et al.: Soil in climate mitigation and adaptation. Proceedings of the National Academy of Sciences 116(21): 10213. doi:10.1073/pnas.1905360116

An L 2012. Modeling human decisions in coupled human and natural systems: review of agent-based models. Ecological Modelling 229: 25–36. doi:10.1016/j.ecolmodel.2011.07.010

Australian Government 2018. Carbon credits (carbon farming initiative – measurement of soil carbon sequestration in agricultural systems) methodology determination 2018. http://www.environment.gov.au/climate-change/government/emissions-reductionfund/methods/measurement-of-soil-carbon-sequestration-in-agricultural-systems.

Balaine N, Clough TJ, Beare MH, Thomas SM, Meenken ED 2016. Soil gas diffusivity controls N_2O and N_2 emissions and their ratio. Soil Science Society of America Journal 80: 529–540.

Baldock JA, Beare MH, Curtin D, Hawke B 2018. Stocks, composition and vulnerability to loss of soil organic carbon predicted using mid-infrared spectroscopy. Soil Research 56(5): 468–480.

Barnett AL, Schipper LA, Taylor A, Balks MR, Mudge PL 2014. Soil C and N contents in a paired survey of dairy and dry stock pastures in New Zealand. Agriculture Ecosystems & Environment 185: 34–40.

Beukes PC, Romera AJ, Gregorini P, Macdonald KA, Glassey CB, Shepherd MA 2017. The performance of an efficient dairy system using a combination of nitrogen leaching mitigation strategies in a variable climate. Science of the Total Environment 599: 1791–1801.

Butterbach-Bahl K, Baggs EM, Dannenmann M, Kiese R, Zechmeister-Boltenstern S 2013. Nitrous oxide emissions from soils: how well do we understand the processes and their controls? Philosophical Transactions of the Royal Society B 368: 20130122.

Chabrillat S, Ben-Dor E, Cierniewski J, Gomez C, Schmid T, van Wesemael B 2019. Imaging spectroscopy for soil mapping and monitoring. Surveys in Geophysics 40(3): 361–399.

Condron LM, Black A, Wakelin SA 2012. Effects of long-term fertiliser inputs on the quantities of organic carbon in a soil profile under irrigated grazed pasture. New Zealand Journal of Agricultural Research 55(2): 161–164.

de Klein CAM, Harvey MJ, Clough TJ, Petersen SO, Chadwick DR, Venterea RT 2020. Global Research Alliance N_2O chamber methodology guidelines: Introduction, with health and safety considerations. Journal of Environmental Quality 49(5): 1073–1080, doi:10.1002/jeq2.20131

Denmead OT 2008. Approaches to measuring fluxes of methane and nitrous oxide between landscapes and the atmosphere. Plant and Soil 309: 5–24.

de Vries FT, Thébault E, Liiri M, Birkhofer K, Tsiafouli MA, Bjørnlund L, Bracht Jørgensen H, Brady MV, Christensen S, de Ruiter PC, et al. 2013. Soil food webs explain ecosystem services. Proceedings of the National Academy of Sciences 110(35): 14296–14301.

Donovan M, Orwin K, Roudier P, Belliss S 2021. Quantifying resilience to drought and flooding in agricultural systems. Manaaki Whenua – Landcare Research Contract Report LC3954-14 for Our Land and Water National Science Challenge & The NEXT Foundation. Downloadable at: https://ourlandandwater.nz/regenag and https://www.landcareresearch.co.nz/publications/regenag

Dowhower SL, Teague WR, Casey KD, Daniel R 2020. Soil greenhouse gas emissions as impacted by soil moisture and temperature under continuous and holistic planned grazing in native tallgrass prairie. Agriculture, Ecosystems and Environment 287: 106647.

Dymond JR 2010. Soil erosion in New Zealand is a net sink of CO_2. Earth Surface Processes and Landforms 35(15): 1763–1772.

FAO 2019a. Global Soil Organic Carbon Map. GLOSIS - GSOCmap (v1.5.0). http://54.229.242.119/GSOCmap/

FAO 2019b. Measuring and modelling soil carbon stocks and stock changes in livestock production systems: guidelines for assessment (Version 1). Livestock Environmental Assessment and Performance (LEAP) Partnership. Rome: FAO. Licence: CC BY-NC-SA 3.0 IGO.

FAO 2020. A protocol for measurement, monitoring, reporting and verification of soil organic carbon in agricultural landscapes – GSOC-MRV Protocol. Rome: FAO. doi:10.4060/cb0509en.

Felber R, Münger A, Neftel A, Ammann C 2015. Eddy covariance methane flux measurements over a grazed pasture: effect of cows as moving point sources. Biogeosciences 12: 3925–3940.

Flesch TK, Wilson JD, Harper LA, Crenna BP, Sharpe RR 2004. Deducing ground-to-air emissions from observed trace gas concentrations: a field trial. Journal of Applied Meteorology 43: 487–502.

Fornara DA, Tilman D 2008. Plant functional composition influences rates of soil carbon and nitrogen accumulation. Journal of Ecology 96(2): 314–322.

Gilbert N 2008. Agent-based models. Quantitative Applications in the Social Sciences. Number 07-153. Los Angeles, CA: Sage.

Gholizadeh A, Žižala D, Saberioon M, Borůvka L 2018. Soil organic carbon and texture retrieving and mapping using proximal, airborne and Sentinel-2 spectral imaging. Remote Sensing of Environment 218: 89–103.

Grace PR, van der Weerden TJ, Rowlings DW, Scheer C, Brunk C, Kiese R, Butterbach-Bahl K, Rees RM, Robertson GP, Skiba UM 2020. Global Research Alliance N2O chamber methodology guidelines: considerations for automated flux measurement. Journal of Environmental Quality 49(5): 1126–1140, doi:10.1002/jeq2.20124

Haque N 2018. Dietary manipulation: a sustainable way to mitigate methane missions from ruminants. Journal of Animal Science and Technology 60: 15.

Harper LA, Denmead OT, Flesch TK 2011. Micrometeorological techniques for measurement of enteric greenhouse gas emissions. Animal Feed Science and Technology 166–167: 227–239.

Hensen A, Skiba U, Famulari D 2013. Low cost and state of the art methods to measure nitrous oxide emissions. Environmental Research Letters 8: 025022.

Hunt JE, Laubach J, Barthel M, Fraser A, Phillips RL 2016. Carbon budgets for an irrigated intensively grazed dairy pasture and an unirrigated winter-grazed pasture. Biogeosciences 13: 2927–2944.

Jackman RH 1964. Accumulation of organic matter in some New Zealand soils under permanent pasture. I. Patterns of change of organic carbon, nitrogen, sulphur and phosphorus. New Zealand Journal of Agricultural Research 7: 445–471.

Jensen JL, Christensen BT, Schjønning P, Watts CW, Munkholm LJ 2018. Converting loss-on-ignition to organic carbon content in arable topsoil: pitfalls and proposed procedure. European Journal of Soil Science 69(4): 604–612.

Johnson K, Huyler M, Westberg H, Lamb B, Zimmerman P 1994. Measurement of methane emissions from ruminant livestock using a SF6 tracer technique. Environmental Science and Technology 28: 359–362.

Jonker A, Farrell L, Scobie D, Dynes R, Edwards G, Hague H, McAuliffe R, Taylor A, Knight T, Waghorn G 2018. Methane and carbon dioxide emissions from lactating dairy cows grazing mature ryegrass/white clover or a diverse pasture comprising ryegrass, legumes and herbs. Animal Production Science 59: 1063–1069.

Jonker A, Molano G, Koolaard J, Muetzel S 2016. Methane emissions from lactating and non-lactating dairy cows and growing cattle fed fresh pasture. Animal Production Science 57: 643–648.

Kirschbaum MUF, Schipper LA, Mudge PL, Rutledge S, Puche NJB, Campbell DI 2017. The trade-offs between milk production and soil organic carbon storage in dairy systems under different management and environmental factors. Science of the Total Environment 577: 61–72.

Lambert MG, Clark DA, Mackay AD, Costall DA 2000. Effects of fertiliser application on nutrient status and organic matter content of hill soils. New Zealand Journal of Agricultural Research 43(1): 127–138.

Lange M, Eisenhauer N, Sierra CA, Bessler H, Engels C, Griffiths RI, Mellado-Vazquez PG, Malik AA, Roy J, Scheu S, et al. 2015. Plant diversity increases soil microbial activity and soil carbon storage. Nature Communications 6: 6707.

Lassey KR 2007. Livestock methane emission: from the individual grazing animal through national inventories to the global methane cycle. Agricultural and Forest Meteorology 142: 120–132.

Laubach J, Grover SPP, Pinares-Patiño CS, Molano G 2014. A micrometeorological technique for detecting small differences in methane emissions from two groups of cattle. Atmospheric Environment 98: 599–606.

Laubach J, Hunt JE 2018. Greenhouse-gas budgets for irrigated dairy pasture and a winter-forage kale crop. Agricultural and Forest Meteorology 258: 117–134.

Laubach J, Hunt JE, Graham SL, Buxton RP, Rogers GND, Mudge PL, Carrick S, Whitehead D, 2019. Irrigation increases forage production of newly established lucerne but enhances net ecosystem carbon losses. Science of the Total Environment 689: 921–936.

Lavallee JM, Soong JL, Cotrufo MF 2020. Conceptualizing soil organic matter into particulate and mineral-associated forms to address global change in the 21st century. Global Change Biology 26(1): 261–273.

Li Z, Kelliher FM 2007. Methane oxidation in freely and poorly drained grassland soils and effects of cattle urine application. Journal of Environmental Quality 36: 1241–1248.

Li Y, Li Z, Chang SX, Cui S, Jagadamma S, Zhang Q, Cai Y 2020. Residue retention promotes soil carbon accumulation in minimum tillage systems: Implications for conservation agriculture. Science of The Total Environment 740: 140–147.

Loisel J, Casellas Connors JP, Hugelius G, Harden JW, Morgan CL 2019. Soils can help mitigate CO_2 emissions, despite the challenges. Proceedings of the National Academy of Sciences May 2019, 116(21): 10211–10212. doi:10.1073/pnas.1900444116.

Luo J, Balvert SF, Wise B, Welten B, Ledgard SF, de Klein CAM, Lindsey S, Judge A 2018. Using alternative forage species to reduce emissions of the greenhouse gas nitrous oxide from cattle urine deposited onto soil. Science of the Total Environment 610-611: 1271–1280.

McMillan AMS, Harvey MJ, Martin RJ, Bromley AM, Evans MJ, Mukherjee S, Laubach J 2014. The detectability of nitrous oxide mitigation efficacy in intensively grazed pastures using a multiple-plot micrometeorological technique. Atmospheric Measurement Techniques 7: 1169–1184.

McNeill SJE, Golubiewski N, Barringer J 2014. Development and calibration of a soil carbon inventory model for New Zealand. Soil Research 52(8): 789–804.

MfE 2020. New Zealand's Greenhouse Gas Inventory 1990–2018. Wellington Ministry for the Environment. https://www.mfe.govt.nz/publications/climate-change/new-zealands-greenhouse-gas-inventory-1990-2018.

Mudge PL 2019. A national soil carbon monitoring system for agricultural land in New Zealand. Soil Horizons Newsletter. https://www.landcareresearch.co.nz/publications/soil-horizons/soil-horizons-articles/national-soil-carbon-monitoring-system-for-agricultural-land/ .

Mudge PL, Kelliher FM, Knight TL, O'Connell D, Fraser S, Schipper LA 2017. Irrigating grazed pasture decreases soil carbon and nitrogen stocks. Global Change Biology 23(2): 945–954.

Mudge PL, McNeill S, Hedley C, Roudier P, Poggio M, Malone B, Baldock J, Smith P, McNally S, Beare M, et al. 2020. Design of an on-farm soil carbon benchmarking and monitoring approach for individual pastoral farms. MPI Technical Paper No: 2020/02.

Muetzel S, Clark H 2015. Methane emissions from sheep fed fresh pasture. New Zealand Journal of Agricultural Research 58: 472–489.

Nazaries L, Murrell JC, Millard P, Baggs L, Singh BK 2013. Methane, microbes and models: fundamental understanding of the soil methane cycle for future predictions. Environmental Microbiology 15: 2395–2417.

Nichols KL, Del Grosso SJ, Derner JD, Follett RF, Archibeque SL, Stewart CE, Paustian KH 2016. Nitrous oxide and methane fluxes from cattle excrement on C3 pasture and C4-dominated steppe. Agriculture, Ecosystems and Environment 225: 104–115.

Niklaus PA, Le Roux X, Poly F, Buchmann N, Scherer-Lorenzen M, Weigelt A, Barnard RL 2016. Plant species diversity affects soil–atmosphere fluxes of methane and nitrous oxide. Oecologia 181: 919–930.

Ogle SM, Alsaker C, Baldock J, Bernoux M, Breidt FJ, McConkey B, Regina K, Vazquez-Amabile GG 2019. Climate and soil characteristics determine where no-till management can store carbon in soils and mitigate greenhouse gas emissions. Scientific Reports 9(1): 11665.

Pijlman J, Berger SJ, Lexmond F, Bloem J, van Groenigen JW, Visser EJW, Erisman JW, van Eekeren N 2020. Can the presence of plantain (Plantago lanceolata L.) improve nitrogen cycling of dairy grassland systems on peat soils? New Zealand Journal of Agricultural Research 63: 106–122.

Pinares-Patiño CS, Gere J, Williams K, Gratton R, Juliarena P, Molano G, MacLean S, Sandoval E, Taylor G, Koolaard J 2012. Extending the collection duration of breath samples for enteric methane emission estimation using the SF 6 tracer technique. Animals 2: 275–287.

Pinares-Patiño CS, Franco FE, Molano G, Kjestrup H, Sandoval E, MacLean S, Battistotti M, Koolaard J, Laubach J 2016. Feed intake and methane emissions from cattle grazing pasture sprayed with canola oil. Livestock Science 184: 7–12.

Project Drawdown 2020. https://www.drawdown.org/solutions

Roque BM, Salwen JK, Kinley R, Kebreab E 2019. Inclusion of *Asparagopsis armata* in lactating dairy cows' diet reduces enteric methane emission by over 50 percent. Journal of Cleaner Production 234: 132–138.

Rousset C, Clough TJ, Grace PR, Rowlings DW, Scheer C 2020. Soil type, bulk density and drainage effects on relative gas diffusivity and N2O emissions. Soil Research, in press, doi:10.1071/SR20161.

Rutledge S, Mudge PL, Wallace DF, Campbell DI, Woodward SL, Wall AM, Schipper LA 2014. CO2 emissions following cultivation of a temperate permanent pasture. Agriculture, Ecosystems and Environment 184: 21–33.

Rutledge S, Wall AM, Mudge PL, Troughton B, Campbell DI, Pronger J, Joshi C, Schipper LA 2017. The carbon balance of temperate grasslands, part I: The impact of increased species diversity. Agriculture, Ecosystems & Environment 239: 310–323.

Saggar S, Hedley CB, Giltrap DL, Lambie SM 2007. Measured and modelled estimates of nitrous oxide emission and methane consumption from a sheep-grazed pasture. Agriculture, Ecosystems and Environment 122: 357–365.

Saggar S, Hedley C, Mackay AD 1997. Partitioning and translocation of photosynthetically fixed 14C in grazed hill pastures. Biology and Fertility of Soils 25(2): 152–158.

Schaufler G, Kitzler B, Schindlbacher A, Skiba U, Sutton MA, Zechmeister-Boltenstern S 2010. Greenhouse gas emissions from European soils under different land use: effects of soil moisture and temperature. European Journal of Soil Science 61: 683–696.

Schipper LA, Dodd MB, Pronger J, Mudge PL, Upsdell MP, Moss RA 2013. Decadal changes in total surface soil C and N in pastures under differing irrigation frequencies and superphosphate fertilizer application rates. Soil Science Society of America Journal 77: 246–256.

Scott JT, Stewart DPC, Metherell AK 2012. Alteration of pasture root carbon turnover in response to superphosphate and irrigation at Winchmore New Zealand. New Zealand Journal of Agricultural Research 55(2): 147–159.

Simon PL, de Klein CAM, Worth W, Rutherford AJ, Dieckow J 2019. The efficacy of Plantago lanceolata for mitigating nitrous oxide emissions from cattle urine patches. Science of the Total Environment 691: 430–441.

Smith KA, Dobbie KE, Ball BC, Bakken LR, Sitaula BK, Hansen S, Brumme R, Borken W, Christensen S, Prieme A, et al. 2000. Oxidation of atmospheric methane in northern European soils: comparison with other ecosystems, and uncertainties in the global terrestrial sink. Global Change Biology 6: 791–803.

Smith P, Bustamante M, Ahammad H, Clark H, Dong H, Elsiddig EA, Haberl H, Harper R, House J, Jafari M, et al. 2014. Agriculture, forestry and other land use (AFOLU). In: Mitigation of climate change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge: Cambridge University Press. https://www.ipcc.ch/report/ar5/wg3/

Smith P, Soussana J-F, Angers D, Schipper LA, Chenu C, Rasse DP, Batjes NH, van Egmond F, McNeill S, Kuhnert M, et al. 2020. How to measure, report and verify soil carbon change to realize the potential of soil carbon sequestration for atmospheric greenhouse gas removal. Global Change Biology 26(1): 219–241.

Sparling GP, Lewis R, Schipper LA, Mudge P, Balks M 2014. Changes in soil total C and N contents at three chronosequences after conversion from plantation pine forest to dairy pasture on a New Zealand Pumice soil. Soil Research 52(1): 38–45.

Stanley PL, Rowntree JE, Beede DK, DeLonge MS, Hamm MW et al. 2018. Impacts of soil carbon sequestration on life cycle greenhouse gas emissions in Midwestern USA beef finishing systems. Agricultural Systems 162: 249–258.

Stenberg B, Viscarra Rossel RA, Mouazen AM, Wetterlind J 2010. Visible and near infrared spectroscopy in soil science. Advances in Agronomy 107: 163–215.

Sun WJ, Canadell JG, Yu LJ, Yu LF, Zhang W, Smith P, Fischer T, Huang Y 2020. Climate drives global soil carbon sequestration and crop yield changes under conservation agriculture. Global Change Biology 26(6): 3325–3335.

Teague R 2020. http://research.agrilife.org/vernon/wp-content/uploads/sites/2/2018/02/Foyer_Project_for_Teague_web_page_.pdf .

Teague WR, Dowhower SL, Baker SA, Haile N, DeLaune PB, Conover DM 2011. Grazing management impacts on vegetation, soil biota and soil chemical, physical and hydrological properties in tall grass prairie. Agriculture, Ecosystems and Environment 141: 310–322.

Thompson LR, Maciel IFC, Cassida K, Rowntree J 2019. Impact of forage diversity on animal production, forage productivity and nutritive value, emissions, and soil C/N cycling. Journal of Animal Science 97(Suppl. S3): 239–240.

Thompson LR, Rowntree JE, 2020. Methane sources, quantification, and mitigation in grazing beef systems. Applied Animal Science 36: 556–573.

van der Weerden TJ, Kelliher FM, de Klein CAM 2012. Influence of pore size distribution and soil water content on nitrous oxide emissions. Soil Research 50: 125–135.

van Groenigen JW, Huygens D, Boeckx P, Kuyper TW, Lubbers IM, Rütting T, Groffman PM 2015. The soil N cycle: new insights and key challenges. Soil 1: 235–256.

Viscarra Rossel RA, Bouma J 2016. Soil sensing: a new paradigm for agriculture. Agricultural Systems 148: 71–74.

Waghorn GC, Jonker A, Macdonald KA 2016. Measuring methane from grazing dairy cows using GreenFeed. Animal Production Science 56: 252–257.

Wall AM, Campbell DI, Mudge PL, Schipper LA 2020. Temperate grazed grassland carbon balances for two adjacent paddocks determined separately from one eddy covariance system. Agricultural and Forest Meteorology 287: 107942.

Wang T, Teague WR, Park SC, Bevers S 2015. GHG mitigation potential of different grazing strategies in the United States Southern Great Plains. Sustainability 7(10): 13500–13521.

Wang ZG, Hoffmann T, Six J, Kaplan JO, Govers G, Doetterl S, Van Oost K 2017. Human-induced erosion has offset one-third of carbon emissions from land cover change. Nature Climate Change 7(5): 345–349.

Ward SE, Smart SM, Quirk H, Tallowin JRB, Mortimer SR, Shiel RS, Wilby A, Bardgett RD 2016. Legacy effects of grassland management on soil carbon to depth. Global Change Biology 22(8): 2929–2938.

Wendt JW, Hauser S 2013. An equivalent soil mass procedure for monitoring soil organic carbon in multiple soil layers. European Journal of Soil Science 64(1): 58–65.