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Measuring Soil Gas Fluxes:

Evaluation of a portable chamber approach to quantify carbon storage in ecological restoration projects, New Zealand

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MASTER THESIS

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Statutory Declaration

I declare that I have written this thesis independently, that I have not used other than the indicated sources, and explicitly marked all material quoted literally or by content from the used sources. Furthermore, I confirm that I have not submitted this thesis elsewhere.

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Abstract

Soils play a critical role in global climate change as they can either act as source or sink of atmospheric greenhouse gases (GHGs). Adapted land management sequestering soil organic carbon (SOC) is an effective tool to mitigate net GHG emissions while simultaneously strengthening food security by improving soil fertility. Thus, there is growing international interest in reliable monitoring of SOC stocks and stock changes. In-situ measurements of soil gas fluxes can deliver valuable information about C and N dynamics on a small scale, and data can be used to improve GHG inventories and SOC models. Even though the accuracy of measurement devices improved significantly, specific expertise and appropriate planning are needed to derive meaningful outcomes. The present thesis aims to evaluate the usability of a portable chamber system for soil gas flux measurements on a farm scale. It assesses how the approach could be implemented into broader restoration projects, delivering complementary data. Therefore, a case study was conducted in an intensively used agricultural area within the Canterbury region, New Zealand, looking at the influence of vegetation complexity and the implementation of trees on soil gas fluxes. Primary outcomes revealed that vegetation composition significantly influenced CO_2 release and that it was lowest under native vegetation dominated by kānuka trees (*Kunzea* sp.). Furthermore, soil moisture was the most influential controlling factor under prevalent dry conditions, limiting CO_2 effluxes even under high temperatures. N_2O effluxes were generally at a low level. Due to methodological difficulties of N_2O measurements under the low natural levels of microbial N_2O production, its use in future projects was finally not recommended. Measurements of CO_2 delivered a meaningful set of data with values comparable to the literature and could thus be effectively implemented in further studies. However, prospective users should pay special attention to a sophisticated experimental design accounting for spatial and temporal variations.

Key words: soil gas fluxes, soil respiration, soil carbon sequestration, greenhouse gas emissions, climate change

Zusammenfassung

Böden spielen eine entscheidende Rolle im globalen Klimawandel, da sie sowohl Quelle als auch Senke atmosphärischer Treibhausgase (THG) sein können. Eine angepasste Landnutzung, die organischen Kohlenstoff im Boden (engl. SOC) bindet, ist ein wirksames Instrument zur Minderung der Netto-THG-Emissionen bei gleichzeitiger Stärkung der Ernährungssicherheit durch eine Verbesserung der Bodenfruchtbarkeit. Daher besteht wachsendes internationales Interesse an zuverlässigem Monitoring von SOC-Beständen und Bestandsveränderungen. In-situ-Messungen von Bodengasflüssen können wertvolle Informationen über kleinskalierte C- und N-Dynamiken liefern; und Daten können zur Verbesserung von THG-Inventaren und SOC-Modellen verwendet werden. Auch wenn sich die Genauigkeit der Messgeräte deutlich verbessert hat, sind Fachkenntnisse und eine sachgemäße Planung erforderlich, um aussagekräftige Ergebnisse zu erzielen. Die vorliegende Thesis hat das Ziel, die Nutzbarkeit eines tragbaren Hauben-Systems zur Messung von Bodengasflüssen zu evaluieren. Es wird bewertet, inwieweit der Ansatz in umfassendere Renaturierungsprojekte integriert werden kann. Dazu wurde eine Fallstudie in einem intensiv genutzten landwirtschaftlichen Gebiet in der Canterbury-Region, Neuseeland, durchgeführt, die den Einfluss der Vegetationskomplexität und der Integration von Bäumen auf Bodengasflüsse untersucht. Die Ergebnisse zeigen, dass die Vegetationszusammensetzung die CO_2-Freisetzung signifikant beeinflusste und dass diese bei einer heimischen, von Kānuka-Bäumen (*Kunzea sp.*) dominierten, Vegetation am geringsten war. Darüber hinaus war die Bodenfeuchte der wichtigste Einflussfaktor unter den vorherrschenden trockenen Bedingungen, der CO_2-Flüsse auch unter hohen Temperaturen limitierte. N_2O-Flüsse waren allgemein gering. Aufgrund methodischer Schwierigkeiten bei N_2O-Messungen unter dem niedrigen Niveau natürlicher mikrobieller N_2O-Produktion, wird von deren Verwendung in zukünftigen Projekten abgeraten. CO_2-Messungen konnten einen aussagekräftigen Datensatz liefern und könnten effektiv in Folgestudien integriert werden. Jedoch sollten zukünftige Nutzer*innen auf ein komplexes Versuchsdesign achten, das sowohl räumliche als auch zeitliche Variabilität einbezieht.

Schlüsselwörter: Bodengasflüsse, Bodenatmung, Bodenkohlenstoff-Sequestrierung, Treibhausgasemissionen, Klimawandel

Abbreviations

AFOLU	Agriculture, forestry, and other land use
AFS	Agroforestry systems
CAP	Common Agricultural Policy
ETS	Emissions Trading Scheme
EU	European Union
GHGs	Greenhouse gases
MRV	Monitoring, reporting and verification
MANOVA	Multifactorial analysis of variance
NDCs	Nationally Determined Contributions
NEE	Net ecosystem exchange
NPP	Net primary productivity
NZ	New Zealand
PKE	Palm kernel expeller
SCMS	Soil Carbon Monitoring System
SCS	Soil carbon sequestration
SOC	Soil organic carbon
SOM	Soil organic matter
SDGs	Sustainable Development Goals
IPCC	The Intergovernmental Panel on Climate Change
UNFCCC	United Nations Framework Convention on Climate Change
WC	Water content

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1 Introduction - Objective and Structure of the Thesis

Agricultural food production is a major driver of global climate change, as soils and livestock emit large quantities of the three major greenhouse gases (GHGs), nitrous oxide (N_2O), methane (CH_4) and carbon dioxide (CO_2), into the atmosphere. However, adapted soil management has also a huge potential to reduce global emissions as soils can act as a natural sink of atmospheric CO_2 (Bossio et al. 2020). Emerging research underlines the critical role of soil organic carbon (SOC) sequestration towards meeting international climate goals (IPCC 2019). Moreover, SOC is a primary indicator of soil fertility and can increase agricultural productivity (Oldfield et al. 2019). Due to the valuable benefits of SOC sequestration for simultaneously strengthening food security and mitigating climate change, there is growing international interest in improved monitoring, reporting, and verification of existing SOC stocks and stock changes (Smith et al. 2020).

In-situ measurements of soil gas fluxes on a field scale are an essential source of data for developing and verifying robust modelling approaches and GHG inventories (Saggar et al. 2008); the use of soil chambers is an inexpensive and easily implementable approach for this (Smith et al. 2010). Field measurements of soil gas fluxes can deliver valuable information about carbon (C) and nitrogen (N) dynamics and help to identify sources and sinks of GHGs. Measurement devices improved significantly in accuracy and usability over the last decades (Smith et al. 2020). However, proper use still needs a certain degree of expertise and experience.

The agricultural landscape of New Zealand (NZ) is dominated by pastoral farming. Sheep and beef farms cover roughly one-third of the countries landmass (9.3 million hectares), mainly on extensively used marginal sites (Beef+Lamb New Zealand 2018). However, the NZ dairy sector enormously expanded and intensified over the last decades. The boom of intensive dairy farms led to significant changes in land use and vegetation composition, resulting in biodiversity losses and increased GHG emissions in large parts of the country (Foote et al. 2015). Increased implementation of sustainable management practices is urgently needed to counteract the adverse environmental effects resulting from the status quo of NZ's agriculture. The inclusion of trees and shrubs into existing pastoral land and its conversion into higher biomass shrublands offers a great potential of sequestering SOC, accompanied by additional biological and economic benefits like provision of new habitats and production of tradable timber (Ramachandran Nair et al. 2010).

The present thesis aims to evaluate the usability of a portable chamber system for soil gas flux measurements on a farm-scale. It thereby intends to assess how far and under which framework conditions the approach can be effectively implemented into broader restoration projects, scientific studies, or agricultural monitoring campaigns that are conducted by personnel with only limited expertise. The method is evaluated through a case study located at two pastoral areas in the Canterbury region, NZ, focussing on the influence of vegetation complexity and the implementation of trees on soil gas fluxes in agricultural systems. Additionally, an extensive literature review shall facilitate the readers' thematic entry into the topic. The thesis is therefore structured as the following:

Firstly, the **state-of-the-art review** will present sound knowledge and exerts of the current research regarding soil C sequestration as a tool to mitigate climate change and strengthen food security. A particular spotlight is on NZ's agriculture and the countries ambitious climate policies.

Secondly, the **practical part** presents the conducted case study. Special attention is paid to methodological aspects. The possibilities and limitations of the soil chamber approach are finally discussed to serve as guidelines for future use.

2 State-of-the-Art Review

2.1 Global Frame

2.1.1 International Commitments

The global climate is changing and thus the living conditions on our planet. Despite long-lasting criticism and doubts, it is extremely likely that anthropogenic activities of the last decades significantly increased the global mean temperature due to an excessive release of GHGs into the atmosphere (IPCC 2013). Besides rising temperatures, natural disasters and extreme weather events are likely to occur more frequently, and the sea level rises due to melting glaciers. If humanity does not succeed to reduce emissions, experts predict wide-ranging consequences for the environment, society, and economy (IPCC 2014). The Intergovernmental Panel on Climate Change (IPCC) assesses the risk of possible climatic changes using computer-based models. These models predict future trends in the global climate and their consequences depending on different development trajectories of humanity. The predictions show clearly that a business as usual scenario could lead to an increase in global mean temperature between 3°C to 5.5°C until 2100, having tremendous impacts on humanity (IPCC 2014).

To limit global warming as well as to adapt to the already irreversible changes of climate change, 154 countries signed the United Nations Framework Convention on Climate Change (UNFCCC) at the Earth Summit, held in Rio de Janeiro, Brazil, in 1992. Besides the UNFCCC the 1992 Earth Summit gave birth to the United Nations Convention to Combat Desertification (UNCCD) and the United Nations Convention on Biological Diversity (UNCBD). All three are often referred to as the three Rio Conventions. To operationalise the UNFCCC, the Kyoto Protocol was adopted in Kyoto, Japan, in December 1997 and entered into force in February 2005, with the 'objective to stabilise atmospheric concentrations of GHGs at a tolerable level, that is not dangerously interfering with the global climatic system'. Each participating country committed to limit or reduce its GHG emissions (UNFCCC 2022). The Kyoto Protocol was followed by the Paris Agreement, adopted at the 21st Conference of the Parties (COP 21) to the UNFCCC in December 2015. Within the Paris Agreement, all participating countries are committing to take actions on climate change, with the aim to reduce global warming to 2°C compared to pre-industrial levels, while pursuing efforts to limit the temperature increase to 1.5°C. Furthermore, it should strengthen participating countries in their resilience to climate change and ensure practical use of financial measures to support the development of low-carbon and climate-resilient economies (Christoff 2016). To deliver towards the goals of the Paris Agreement all participating countries developed and regularly update their Nationally Determined Contributions (NDCs) that must be submitted to the UNFCCC.

The particular importance of land ecosystems for climate change mitigation and adaptation was highlighted by the IPCC Special Report on Climate Change and Land, published in 2019 (IPCC 2019). The report revealed that around 23% of net anthropogenic GHG emissions are attributable to agriculture, forestry, and other land use (AFOLU). At the same time, it showed that the land biosphere acts as a natural net sink for nearly 30% of anthropogenic CO_2 emissions, which is taken up by vegetation and soils. However, this land sink is vulnerable to climate change impacts and other environmental and human pressures. Following these findings, most countries are already integrating the agricultural sector in their NDCs (Ross et al. 2019). Alongside the Paris Agreement in 2015, the 4 per 1000 Initiative was founded. It aims to increase SOC sequestration by adapting agricultural practices to local environmental, social, and economic conditions (Minasny et al. 2017).

Challenges arising through climate change are also addressed in the global Agenda 2030 with its Sustainable Development Goals (SDGs). The goal of global food security - in fulfilment of SDG 2 (No Hunger) - implies increased and adapted agriculture to the changing climatic conditions. At the same time, agriculture will have to contribute to achieving SDG 13 (Measures for climate protection) and SDG 15 (Life on land). Soils play a particular role within the SDGs (Keesstra et al. 2016).

The urgency to mitigate climate change and adapt to already irreversible changes led several countries to develop strategies for their transformation towards climate-neutral societies. One example here is the 'European Green Deal', presented in December 2019, with which the European Union (EU) aims to reach climate neutrality by 2050 (European Commission 2019). As the agricultural sector is responsible for 10.3% of the EU's GHG emissions, implementing climate and environmentally friendly agriculture has a high priority (European Commission 2020). This priority is also made clear by the European Farm to Fork Strategy (European Commission 2020) and the environmental and climate-friendly orientation of the new Common Agricultural Policy (CAP), proposed in June 2018 (European Commission 2018). One example of a new green business model within the Farm to Fork Strategy is the sequestration of C by farmers and foresters. Management practices that remove CO_2 from the atmosphere shall be rewarded through the new CAP or other incentives, such as carbon markets. This new business model is promoted by the novel EU carbon farming initiative under the Climate Pact and will offer farmers a new source of income while supporting the decarbonisation of food chains. To monitor and verify the validity of C removals, the EU is developing a regulatory framework based on robust and transparent methods for C accounting (European Commission 2020).

The importance of preserving SOC stocks to fulfil the EU's emission reduction commitments is addressed in the EU Soil Thematic Strategy (European Commission 2012). The Strategy is currently updated, and the 'New Soil Strategy – healthy soil for a healthy life' is supposed to be published in the second quarter of 2021. The associated roadmap, which aims to inform citizens and stakeholders about the Commission's work, highlights the conservation and restoration of soil carbon stocks as a top priority to implement the Green Deal and deliver on international climate commitments. Another pioneer in striving towards climate neutrality is New Zealand, which adopted its 'Zero Carbon Act' in November 2019 (Ministry for the Environment 2019a). NZ's climate strategy will be further examined in one of the following chapters.

As mentioned above, robust monitoring, reporting and verification (MRV) systems of GHG emissions (Singh et al. 2016) and SOC stocks (Smith et al. 2020) are crucial to report on climate change mitigation goals reliably. GHG-inventories from local to national (or even global) scale are based on mathematical models combined with spatial data and data derived from direct measurements. Besides direct measurements of SOC, different novel approaches to estimate C stock changes and GHG emissions from soils become widely used. They reach from flux measurements to non-destructive field-based spectroscopic methods or estimations C stock changes through remote sensing (Smith et al. 2020). As flux measurements using manual chambers are a substantial part of this thesis, in-depth explanation and considerations regarding the strengths and weaknesses of the method will be discussed at a later point.

An excellent example of the use of extensive soil gas flux measurements in national GHG-inventories is the novel approach of Germany for MRV of anthropogenic GHG emissions from drained organic soils (Tiemeyer et al. 2020). The approach is based on flux data of CO_2, N_2O and CH_4 from a national data set compromising more than 250 annual GHG balances from 118 sites of various land-use categories. These measurements were performed with harmonised protocols using manual soil chambers (Tiemeyer et al. 2020).

Besides its importance for reliable GHG inventories, sound MRV systems for SOC stocks and stock changes are likely to become more critical in the upcoming years to make SOC sequestration efforts more popular within (voluntary) carbon markets. At present, there are only two recognised standards for soil C projects, namely the Verra standard (*verra.org*) and the Gold standard (*goldstandard.org*). However, carbon markets are still in their infancy and likely to become more prevalent in future low carbon societies.

2.1.2 Agriculture and Climate Change

Accounting for 21-37% of global GHG emissions, current food and land use systems are among the main drivers of climate change. Some 9-14% of global emissions come directly from agriculture. The most emission-intensive components of agriculture are animal husbandry, the use of mineral fertilisers and wet rice cultivation. A 5-14% are attributable to land use and changes in land use (e.g. converting forests and wetlands into arable land and pastures). Roughly 5-10% of emissions are generated along agricultural supply chains, for example, during storage, processing, and transportation of agricultural products (Jia et al. 2019). Table 2-1 compares total emissions and shares to overall emissions from the food system components Agriculture (crop and livestock activities within the farm gate), Land Use (land use and land-use change dynamics associated with agriculture) and Beyond Farm Gate (food processing, retail, and consumption patterns, including upstream and downstream processes such as the manufacture of chemical fertilisers and fuel).

Growing consumption of animal proteins and rising food waste, especially in industrialised nations and emerging economies, also significantly impact the emissions footprint (Jia et al. 2019). The contribution of food systems to overall emissions is likely to increase in the future and could account for up to 70% of total allowable GHG emissions from anthropogenic sources by 2050 (IPCC 2014a).

Table 2-1: GHG emissions ($GtCO_2$-eq year^{-1}) from the food system and their contribution (%) to total anthropogenic emissions. Mean of 2007-2016 period. (adapted from IPCC 2019)

Food system component	**Emissions (Gt CO_2-eq year^{-1})**	**Share in mean total emissions (%)**
Agriculture	6.2 ±1.4	9-14
Land use	4.9 ±2.5	5-14
Beyond farm gate	2.6 ±5.2	5-10
Food systems (total)	10.8 ±19.1	21-37

Yet, agriculture is not only a significant driver of climate change; it is also severely suffering under its impacts. Agricultural production in developing countries is most affected. Adverse effects of climate change on agriculture include increased average annual temperature, changes in water availability (drought or flooding), increases in the number and duration of extreme weather events, and increased pressure from pests and diseases (Global Commission on Adaptation 2019). Especially, heavy rainfall and increased temperatures will further intensify and accelerate soil degradation and thus increase the vulnerability of agroecosystems to periods of drought. Vast areas of arable land are lost annually to soil erosion or have to be abandoned due to salinisation and alkalisation, which are promoted by irrigation practices (Pimentel and Burgess 2013).

Climate change was shown to have adverse physiological effects on crops and livestock (Rötter and Van De Geijn 1999). Rising temperatures reduce water availability for crops by drying out air and soil. Furthermore, high temperatures suppress livestock productivity and increase the pressure of parasites and diseases on animals (Simpkin et al. 2020). Elevated concentrations of CO_2 in the atmosphere could potentially benefit plant growth and offset some of the negative effects. However, such CO_2-fertilization effects are uncertain and yet only poorly understood. Some research even indicates that increasing CO_2 concentrations could reduce the nutritional quality of many crops (Zhu et al. 2018).

The effects of climate change will undoubtedly intensify over the upcoming decades due to current and near-term emissions. Even if humanity would successfully meet its target to limit the global average temperature increase to well below 2°C, as defined in the Paris Agreement, impacts will still be visible, affecting some regions more than others. Thus, agricultural systems must become more resilient towards climate-related risks and drastically mitigate GHG emissions simultaneously.

2.2 Soils and Climate

Soils and climate are strongly interconnected. Solar energy is partly absorbed and partly reflected by the earth surface. Absorbance depends on the heat capacity and heat conductivity, while reflection, also called albedo, depends on the colour and cover of the surface. The amount of solar energy that retains in the atmosphere is influenced by the atmospheric concentration of radiatively active gases (CO_2, CH_4, N_2O, O_3, H_2O), generating the *greenhouse effect*. Soils substantially influence the atmospheric concentration of these GHGs, either acting as a net source or sink (Figure 2-1).

Climatic parameters that strongly impact soils are temperature, water, gaseous composition, and nutrients. Thus, climate change directly affects soil processes such as decomposition, erosion, weathering, desertification, and salinisation. Other indirect impacts occur through climatic effects on net primary productivity (NPP), hydrological cycles and energy balance (Lal 2019). Soil formation is mainly affected by prevailing temperature and moisture, influencing the weathering process of parent material and soil genesis. In wet climate regions, soils are often leached of basic cations leading to elevated acidity and high concentrations of Al, Mn, and Fe, resulting in toxicity and inhibited plant growth. Warm and dry regions have higher NPP, influencing soil processes through higher biomass C inputs (Lal 2019). The hydrothermal regime also affects soil biota, influencing the rate of C decomposition and the release of CO_2 (Hartemink and Mcsweeney 2014). Elevated levels of CO_2 in the atmosphere could enhance NPP through CO_2 fertilisation effects. However, other factors like N and water availability will mostly be limiting factors (Reich et al. 2014).

Conversely, soils also impact climate, as the coupled cycling of H_2O, C, N, and P shape the atmospheric chemistry. The strong interlinkages of soil, climate, and vegetation become particularly evident looking at soil moisture, constraining photosynthesis, and transpiration of plants. Changes in evaporation, in turn, can have cooling effects on temperature (Seneviratne et al. 2010). The coupled C cycle of soil and atmosphere and the resulting ability of soils to act as a natural C sink was already pointed out in the previous chapter. Soil degradation is depleting C stocks, while restoration measures can increase C storage, reducing atmospheric CO_2 concentrations. Besides the impact of soil C on GHG emission, SOC content also influences soil albedo through changes in soil colour (Lal 2019). This relationship makes the moderation of climate an essential natural ecosystem service of soils. The interrelationship between managed and unmanaged ecosystems with climate is illustrated in Figure 2-1.

Serious climatic threats arise through the thawing of frozen soils (Cryosols) in the arctic tundra and alpine regions that store huge SOC reserves under their ice layers. Rising temperatures will lead to rapid decomposition and accelerate methanogenesis processes resulting in higher emissions of CO_2 and CH_4 into the atmosphere. Additionally, soil N could also be released as N_2O through O_2 induced nitrification and denitrification processes (Lal 2013). A similar relevance regarding global climate is attributed to wetlands. They only cover 3% of the land surface but store roughly 25% of the worlds SOC, making it the most effective intact C storage on earth (Sachs et al. 2015). Desiccation or drainage of these soils releases significant parts of the stored C as CO_2 into the atmosphere.

Based on the temperature sensitivity of SOC decomposition, rising temperatures could finally create a feedback loop; a warming climate increases decomposition leading to enhanced GHG emissions, which in turn cause rising temperatures. This feedback loop could accelerate global warming above usual predictions. The interrelation between climate change and the terrestrial C cycle is yet not fully understood. Further clarification through research will be crucial to develop adapted measures, increasing the capability of soils to sequester C in the long run. The upcoming chapter will delve into the current understandings of C dynamics in soils.

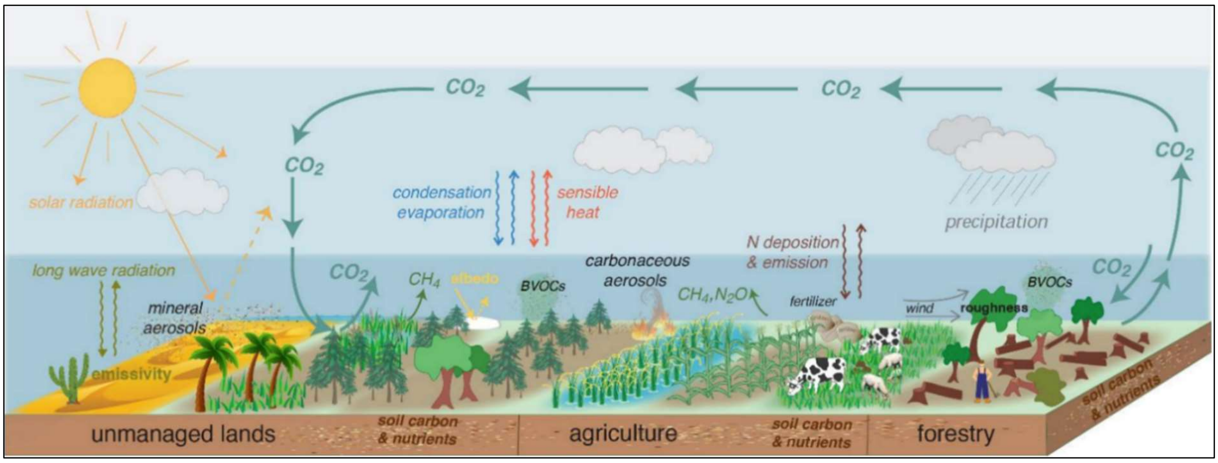

Figure 2-1: The structure and functioning of managed and unmanaged ecosystems that affect local regional and global climate. (IPCC 2019) - Land surface determines absorption and reflection of solar radiation. Land ecosystems emit or remove GHGs and precursors of short-lived climate forcers such as biogenic volatile organic compounds (BVOCs) and mineral dust. Aerosols formed from these precursors affect regional climate by altering the amounts of precipitation and radiation reaching the land surfaces.

2.3 Carbon in Soils

C in soils represents a substantial part of all terrestrial biomes. The world's soils contain approximately 1500 Pg (1 Pg = 1 Gt = 10^{15} g) of organic C in the top metre (Kutsch et al. 2012), roughly three times the amount of C in vegetation and twice the amount in the atmosphere (IPCC 2014). Already small changes in overall SOC can significantly alter the global carbon cycle, influencing atmospheric CO_2 concentration, as described in the previous paragraphs. Thus, soil C became an essential topic in the context of anthropogenic GHGs and climate change, and the UNFCCC obliges participating countries to provide an accurate reporting of CO_2 and other GHG emissions, containing data on major national sinks and sources of C.

SOC is also a major indicator for soil fertility and, thus, for its agricultural productivity. SOC positively influences several chemical, physical, and thermal soil properties while promoting biological activity. It thereby increases the water holding capacity of soils, decreasing the need for irrigation. Improvements of the micro-aggregate structure stabilise soils and prevent erosion. The elevation of the cation exchange capacity diminishes the risk of nutrient losses through leaching (Ardö and Olsson 2003).

The initial **source of organic C in soils** is the photosynthetic uptake from atmospheric CO_2 through plants. CO_2 taken up by plants is either converted into biomass (primary production) or directly transported to the roots via the phloem as assimilates. There it is respired by the roots, their mycorrhizal symbionts, and the microbiota of the rhizosphere. Dead plant material is transferred to the soil as litter, where it is decomposed by soil microorganisms, including bacteria, fungi, and meso- and macrofauna. Microbial respiration releases a part of the C in the form of CO_2 back into the atmosphere. At the same time, the litter is converted into stable organic humus forms - a process called *C sequestration* (Kutsch et al. 2012). Microbial C turnover is referred to as *heterotrophic respiration*. The release of CO_2 from plant roots is referred to as *autotrophic respiration* (Hanson et al. 2000). Both processes build up the *ecosystem respiration*, the overall rate of CO_2 efflux from soils to atmosphere. The difference between the photosynthetic uptake of C and the respirational efflux gives the net ecosystem exchange (NEE). A good understanding of the biological turnover of SOC is crucial to understand C dynamics and the C storage capacity of soils. Still, it is highly complex, as autotrophic and heterotrophic respiration are hardly separable (Kutsch et al. 2012).

The **CO_2 flux between soil and atmosphere** can be measured using a variety of gas analysing systems. Measurements on a smaller scale are usually conducted using flux measurement chambers; distinctions are made between open and closed chambers. Closed chambers determine the CO_2 efflux based on the concentration increase within the chamber headspace during a specific period. In contrast, open chambers (or steady-state-flow-through chambers) derive the CO_2 efflux from the concentration difference of the inlet and the outlet of the chamber (Norman et al. 1997). The measurement approach used in the context of this thesis is a closed chamber approach, further described in the Material and Method section. For large scale estimations of NEE, flux towers based on the eddy covariance technique are most used, often in combination with remote sensing or land surface models (Wang et al. 2016). The following paragraphs will set a thematic foundation regarding commonly used terms and critical processes around soil C dynamics:

The term soil organic carbon (SOC) is generally used to describe all C in soils derived from organic origin. Apart from SOC, there is soil inorganic carbon (SIC), primarily calcium (and magnesium) carbonates. Soil organic matter (SOM) is another term often used in the context of soil C. It describes a mixture of materials, mostly particulate organics, humus and charcoal, together with living and dead microbial biomass and fine plant roots. SOM is widely agreed to contain about 58% SOC (Stockmann et al. 2013). Within this thesis, the term soil respiration is used to describe the production of CO_2 by organisms and plant parts in the soil, as defined by Luo and Zhou (2006, p. 5). It is sometimes used synonymously with soil CO_2 efflux.

Soil carbon sequestration (SCS) occurs when atmospheric CO_2 is transferred into recalcitrant soil C-pools, preventing rapid microbial decomposition. A sequestration timeframe of 100 years is usually considered a permanent increase (Lorenz and Lal 2014). The sequestration rate of soils is mainly determined by their texture and structure, prevailing rainfall and temperature, and farming system and soil management (Lal 2004). After changes in land-use or agricultural practices promoting SCS, SOC stocks generally reach their maximum rate of increase between 5-20 years after adoption. Following a sigmoid curve, C levels can increase until saturation is reached (Lal 2004). The theoretical maximum amount of C, a soil can sequester under optimal conditions and independent of climate, management practices and rates of C input is referred to as carbon stabilisation capacity (Castellano et al. 2015). Most soils used for agricultural purposes are depleted in SOC. Thus, their C stabilisation capacity is not yet reached, and they can stabilise additional C under adapted management. Lal (2004a) estimated the cumulative historic soil C loss of managed ecosystems at 55 to 78 Gt. This value, in turn, approximately equals the potential capacity of these soils for SCS. Even though the attainable capacity to stabilise C is usually only 50 – 66% of the potential capacity, ambitious sequestration efforts in agriculture would still significantly affect global C dynamics (Lal 2004a).

Traditionally, the **decomposition of SOC** was explained using the *Humus Concept*, which assumed that long-lived (recalcitrant) SOC is formed from organic compounds that are inherently resistant to decomposition. The concept got questioned as these chemically recalcitrant compounds (humic substances) could not be observed in-situ using modern analytical techniques (Lehmann and Kleber 2015). Findings instead suggested that humic substances are mostly a product of the chemical extraction methods used, rather than a real component of the organic matter (Hartemink and Mcsweeney 2014). According to current understandings, the chemical composition of input material can only describe the short-term decomposition of SOC. Instead, biological and environmental conditions were found to have a more considerable influence in the long run (Amelung et al. 2008). The recalcitrance of SOC is thus not determined by its intrinsic properties but by an interplay of physicochemical and biological factors, influencing the decomposition process (Schmidt et al. 2011). The main factors influencing the decomposition rate are moisture and temperature, decomposability of the material (litter quality), and the compositions of the microbial community (Stockmann et al. 2013). Generally, bacteria use more simple organic compounds like root exudates or fresh plant residues, while fungi use more complex compounds like fibrous plant residues, wood or soil humus (Hartemink and Mcsweeney 2014).

Turnover through root- and rhizosphere respiration is considered as rapid C cycling, usually taking days to months. The decomposition of litter, in turn, ranges in time frames of months to years. Decomposition of inherent (native) SOM seems to be the slowest, with turnover rates of decades to centuries (Stockmann et al. 2013) (Table 2-2). Strong short-term effects on the turnover of SOM were reported after the input of fresh organic material. This labile and easily degradable C can stimulate the activity of microorganisms, leading to accelerated microbial enzyme production connected to increased decomposition rates; this process is often referred to as *priming-effect* (Kuzyakov et al. 2000; Cheng et al. 2017).

Table 2-2: Forms of (soil) organic C found in scientific literature. (adapted from Stockmann et al. 2013)

Form	Composition	Pool category
Surface plant residue	Plant material residing on the surface of the soil, including leaf litter and crop/pasture material	**Fast (or labile) pool** Decomposition occurs at a timescale of days to years
Buried plant residues	Plant material greater than 2 mm in size residing within the soil	**Fast (or labile) pool** Decomposition occurs at a timescale of days to years
Particulate organic matter (POC)	Semi-decomposed organic material smaller than 2 mm and greater than 50 µm in size	**Fast (or labile) pool** Decomposition occurs at a timescale of days to years
Mineral-associated OC ('Humus')	Well decomposed organic material smaller than 50 µm in size that is associated with soil particles	**Slow (or stable) pool** Decomposition occurs at timescale of years to decades
Resistant organic carbon (ROC)	Charcoal or charred materials that results from the burning of organic matter (resistant to biological decomposition)	**Passive (or recalcitrant) pool** Decomposition occurs at a timescale of decades to millennia

Under the new emerging paradigm, the extent of physical protection of soil C is considered one of the main factors influencing its long-term recalcitrance. It determines the accessibility of organic material to microbial decomposers (Kleber et al. 2011; Kallenbach et al. 2016). Regarding the physical protection of C, a significant role is played by the interaction between SOC and the clay fraction, forming organo-mineral complexes or stable aggregates. The encapsulation of SOC within stable micro-aggregates is protecting it against microbial decomposition. Another critical aspect of aggregate formation and SOC dynamics is the fungal colonisation of particulate organic matter (Lal 2013).

SOC dynamics represent a large source of uncertainty when investigating biogeochemical interactions of terrestrial ecosystems with atmosphere and climate. Emerging research leads to new understandings and reconsiderations regarding the C cycle and the size of SOC pools, especially with depth (Jia et al. 2019). Deep soil layers seem to contain much more C than previously thought (González-Jaramillo et al. 2016), and deep SOC was estimated to be very old, with residence times of several thousand years, using radiocarbon measurements (Rumpel and Kögel-Knabner 2011). However, the dynamics of deep SOC remain uncertain, and it is not included in most studies and C models. Nevertheless, some research suggests that warming could significantly increase the CO_2 release from deep soil layers. Hicks Pries et al. (2017) estimated in a deep warming experiment on mineral soils that a 4°C temperature increase would enhance overall annual soil respiration by 34 – 37%.

2.4 Potential of Soils as Carbon Sink

Estimations in the scientific literature on the global GHG mitigation potential through SCS in mineral soils vary widely, ranging from 0.4 to 8.1 Gt CO_2-eq per year, depending on an interplay of various factors, like past and current land management, soil type, resources availability, environmental conditions and microbial composition (Jia et al. 2019). As soils are a finite C sink, sequestration rates decline to insignificant levels as C saturation is reached, often happening within a couple of decades (Smith and Dukes 2013). It is therefore advisable to focus SCS efforts primarily on C depleted areas.

The **4 per 1000 initiative**, launched alongside the Paris Agreement in 2015, is a prominent example of global endeavours to use soils as a natural C sink. The initiative aims to increase global SOM stocks by 4 per 1000 (0.4%) per year to compensate for anthropogenic GHG emissions. The 0.4% goal is based on the complete global soil profile. However, agricultural soils have generally the highest potential to sequester C, especially those with low initial levels of C. According to Minasny et al. (2017), a global increase of 0.4% in C to a depth of 1 m, only on agricultural soils, could possibly sequester between 2-3 Gt C per year and thus offset 20-35% of global anthropogenic GHG emissions. A recent study by Bossio et al. (2020) finds that a strong upscaling of techniques increasing soil C could remove up to 5.5 Gt of CO_2 from the atmosphere. However, the study emphasises that this can only be achieved by good communication and collaboration between farmers, policymakers, and marketeers.

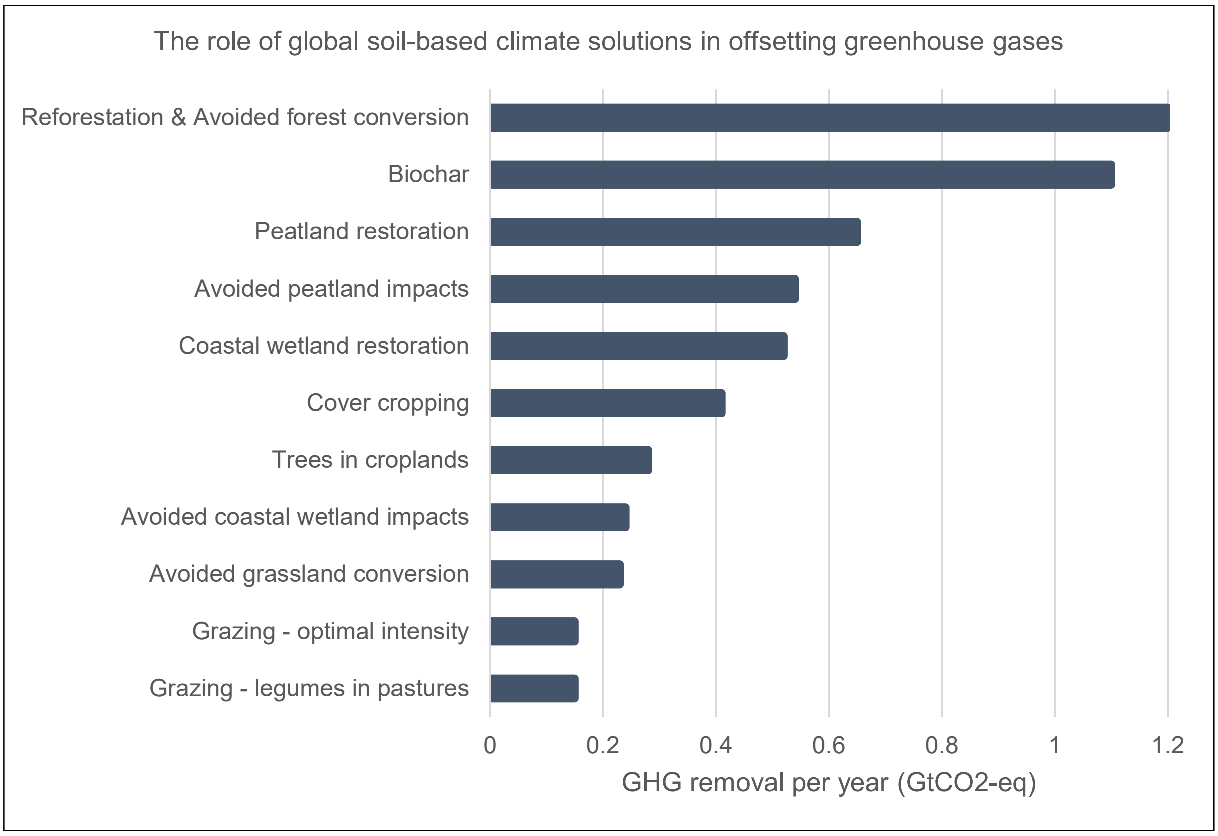

Figure 2-2: The global greenhouse gas removal potential of various soil-based natural climate solutions. Data source: Bossio et al. (2020)

Most significant changes in SOC stocks occur through changes in land use. Bossio et al. (2020) identified the most effective soil-based climate solutions to be reforestation/ avoided forest conversion, peatland restoration/ avoided peatland impact and the use of biochar (Figure-2-2). Conversely, the transformation of natural ecosystems (forests and peatlands in particular) to agricultural areas releases vast amounts of CO_2 into the atmosphere. Agricultural grassland conversion also significantly influences C dynamics. A meta-analysis by Poeplau and Don (2015) showed that grasslands lose about 36% of their initial SOC stocks after 20 years when converted to cropland.

Generally, land use changes from less complex to more complex ecosystems are likely increase SOC stocks. Diversification of agricultural landscapes is therefore almost always preferable to monocultural farming (Paustian et al. 2016).

2.5 Agricultural Practices to Sequester Carbon - Agroforestry

Agricultural management practices increasing SOC stocks are diverse. They cover, among other things, soil and ecosystem restoration, minimum and no-till soil cultivation, use of crop rotations and cover crops, efficient nutrient management including the use of organic fertilisers like manure and compost, efficient irrigation, use of soil amendments like biochar, and agroforestry (Lal 2004). Generally, management practices adding high amounts of biomass while minimising soil disturbance increase SOC by improving soil structure and support of soil organisms. When estimating the offset of GHG emission through SCS measures, it is essential to also consider side effects on the N cycle, microbial responses, and thus the creation of N_2O (Paustian et al. 2016). Li et al. (2005) estimated the effects of SCS measures like reduced tillage and crop residue or manure recycling on N_2O emissions to offset large parts of the sequestered C in terms of CO_2 equivalence. One promising approach to increase SCS in agriculture is agroforestry, a concept that will be further examined in the following paragraphs:

The Association for Temperate Agroforestry (AFTA) defines **agroforestry** as 'an intensive land-management system that optimises the benefits from the biological interactions created when trees and/or shrubs are deliberately combined with crops and/or livestock.' (AFTA 2021). Possible benefits are diverse and go far beyond SCS. They include biodiversity conservation and cropland protection, provision of food and feed for humans and animals, support of pollen transporting insects, and production of wood and timber (Abbas et al. 2017). Moreover, agroforestry systems (AFS) were shown to significantly reduce water losses through drainage and evaporation from soil surfaces, improving overall water use efficiency (Bayala and Wallace 2015).

A large and growing body of literature has investigated the positive effects of AFS on C storage in above and below-ground biomass and long-term SCS. Several relevant studies are reviewed by Abbas et al. (2017) and evaluated as a meta-analysis by De Stefano and Jacobson (2018). Both identify agroforestry as an effective climate-smart agricultural practice. The Kyoto Protocol also recognises the importance of afforestation and reforestation practices to increase C stocks of terrestrial ecosystems. It includes AFS as a particularly efficient practice linking agriculture with afforestation efforts. Moreover, the IPCC (2018) emphasises agroforestry as a powerful combination of climate change adaptation and mitigation. According to Kumar and Nair (2011) estimations, 630 million ha of unproductive cropland and grasslands could be converted globally to AFS, resulting in an overall C sequestration potential of 0.586 Tg C per year by 2040 ($1\ Tg = 10^{12}\ g$).

Below ground C sequestration is very inhomogeneous in space. Higher amounts of C are stored in deeper soil layers, and SOC is often concentrated around the trees. Moreover, species richness and tree density were found to be positively correlating with SOC contents (De Stefano and Jacobson 2018). Still, a major challenge of implementing AFS as a long-term C sequestration strategy is the inconsistency in available datasets arising through differences in methodologies. Biomass and soil C are mostly equated when calculating total sequestered C, a simplification that is often leading to misinterpretations. Furthermore, extrapolation of measured data is highly complex, easily resulting in spatial errors (Nair 2011). Thus, scientific methods have to become more precise and comparable to assess realistic effects.

2.6 Soil N_2O Emissions and Agriculture

The global nitrogen (N) cycle has high importance for our economy, food security, and the functioning of ecosystems. On the other hand, human interventions exert a strong influence on it. Nevertheless, the N-cycle receives relatively little public attention compared to the C-cycle and its impact on atmospheric CO_2 and CH_4 concentrations. Microbial denitrification and nitrification processes lead to emissions of N_2O into the atmosphere. N_2O is a potent GHG with a warming potential 298 times higher than CO_2. It furthermore reacts with stratospheric ozone, damaging the atmosphere's protective layer against incoming UV radiation (Ravishankara et al. 2009).

Anthropogenic N_2O emissions rapidly increased since the industrial revolution, from an atmospheric concentration of 260 ppb in 1850 to current concentrations of 320 ppb (Forster et al. 2007), from which 56-70% can be attributed to agricultural activities (Butterbach-Bahl and Dannenmann 2011). One important reason for the steep increase of agricultural N_2O emissions was the development of the Haber Bosch process, allowing the industrial fixation of atmospheric N_2 into ammonia (NH_3) and leading to a constantly increasing global use of N-fertilizers to sustain prevailing agricultural production systems (Fowler et al. 2013). The excessive use of N-fertilizers in agriculture is related to severe environmental problems. Causing, besides N_2O-emission, pollution of ground and surface waters through nitrate leaching (Byrnes 1990).

N_2O is a by-product of the microbial transformation of N through processes of denitrification and nitrification. Nitrification refers to the enzymatic oxidation of ammonium (NH_4^+) to nitrate (NO_3^-) through specific nitrifying microorganisms, involving hydroxylamine (NH_2OH) and nitrite (NO_2^-) as intermediate products (Equation 2-1). It is a major pathway of N flows into agricultural systems. Nitrification can be divided into two steps, ammonia oxidation and nitrite oxidation. It is primarily determined by ammonia and oxygen availability in the soil. NO or N_2O formation can occur if conditions are unfavourable for further oxidation to NO_3^- (Canfield et al. 2010; Zechmeister-Boltenstern et al. 2019). Denitrification is the reduction of NO_3^- to the N-gases NO, N_2O and N_2 (Equation 2-2). It is a crucial part of the N-cycle, as it returns N from soils into the atmosphere. Denitrification processes occur only under oxygen limitation, so under unfavourable conditions for the aerobic degradation of SOM, using NO_2^- instead of O_2 as an electron acceptor. Besides anaerobic conditions, denitrification is favoured by a high concentration of N substrate (especially NO_3^-) and organic C as an energy source. Incomplete denitrification can result in the formation of N_2O as an end-product. Emissions are especially high from fertilised agricultural soils (Zechmeister-Boltenstern et al. 2019).

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + H_2O + 2H^+$$

Equation 2-1: Nitrification

$$4NO_3^- + 4H^+ + 5C_{org} \rightarrow 5CO_2 + 2N_2 + 2H_2O$$

Equation 2-2: Denitrification

The microbial formation or consumption of N_2O is influenced by several environmental factors, such as soil temperature and moisture. Soil moisture is a critical factor because it determines oxygen availability and soil redox potential (Butterbach-Bahl et al. 2013). Land-use type and management practices also strongly influence N_2O emissions, on the one hand through the amount of N inputs (e.g. fertiliser, amendments), and on the other hand through N-outputs (e.g. grazing, harvesting). Moreover, the predominant vegetation cover significantly determines microbial processes in the soil (Schaufler et al. 2010).

2.7 Soil CH_4 Fluxes

Methane (CH_4) is, after CO_2, the second most abundant GHG in the atmosphere. Even though it is 200 times less abundant than CO_2, its global warming potential is about 30 times higher, making CH_4 a primary contributor to global warming (Wuebbles 2002). Ice core data revealed that atmospheric CH_4 concentrations were at a constant level of 0.7 ppm over the last two millennia but had more than doubled to 1.8 ppm since preindustrial times (Crutzen and Lelieveld 2001). Concentrations are likely to further increase in the upcoming decades, accelerated by CH_4 feedbacks to a warming climate (Dean et al. 2018).

The removal of CH_4 from the atmosphere happens mainly through its chemical destruction in the troposphere by photochemical oxidation with OH radicals, accounting for more than 87% of total atmospheric CH_4 removal (only 7% is removed in the stratosphere) (Lelieveld et al. 1998). This process explains the relatively short mean residence time of CH_4 in the atmosphere of only 12 years (IPCC 2014a). CH_4 plays an important role in the global C-cycle. Its C-atom is fully reduced and cannot accept further electrons. However, it is a potential electron donor. These chemical properties are contrasted by CO_2, the most oxidised form of C, which cannot donate electrons but is a potential electron acceptor. All intermediate forms of C in the cycle have a redox potential between CH_4 and CO_2 (Topp and Pattey 1997). Emissions of CH_4 can have natural and anthropogenic sources. However, the largest emissions originate from microbial sources. Primary natural sources of CH_4 include enteric fermentation in ruminants and some insects (e.g. termites), and anaerobic decomposition of organic matter in soils (Topp and Pattey 1997).

CH_4 in soils is produced as a result of the anaerobic decomposition of organic matter through microbial methanogenesis. Methanogenic archaea convert acetate (acetotrophic methanogens, Equation 2-3) or rather H_2 and CO_2 (hydrogenotrophic methanogens, Equation 2-4) to CH_4. This conversion is the last step in the anoxic food chain, occurring in most anaerobic environments (Zechmeister-Boltenstern et al. 2019).

$$CH_3COO^- + H_2O \rightarrow CH_4 + HCO_3^-$$

Equation 2-3: Acetotrophic methanogenesis

$$CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O$$

Equation 2-4: Hydrogenotrophic methanogenesis

Methanogenic archaea are limited to a number of substrates as their energy source. Thus, most organic compounds (e.g., fatty acids and carbohydrates) must be degraded first by other anaerobic microorganisms, building cooperations in the anaerobic food chain. Methanogenesis is only taking place under highly reduced conditions in the absence of other electron acceptors like nitrate, sulfate or ferric ions.

Soil CH_4 is mainly emitted from wetlands, landfills, and rice paddies. Together these sources are estimated to contribute to about half of global CH_4 emissions (Zechmeister-Boltenstern et al. 2019). While CH_4 is stable in the absence of O_2, under aerobic conditions, it mineralises to CO_2. The oxidation in soils happens through methanotrophic bacteria that live at the oxic-anoxic interface of environments. These aerobic bacteria can cover their energy and C demand through the oxidation process. They feed on the CH_4 produced by methanogens and prevent a large part of CH_4 emissions into the atmosphere (Hanson and Hanson 1996). Reeburgh (2013) estimated the CH_4 consumption of methanotrophic bacteria in affected areas at about 42% of the total CH_4 produced. Whether a soil is a net source or sink of CH_4 mainly depends on the relative rates of methanogenic and methanotrophic bacteria. Both processes are influenced by physicochemical soil properties, primarily moisture, temperature, and pH (Topp and Pattey 1997). A study by Dalal and Allen (2008) compared GHG emissions from unmanaged ecosystems. They concluded that most forests are a net sink for CH_4, while tundra and wetlands are significant sources of natural CH_4 emissions.

2.8 Emissions of the Livestock Sector – Carbon Sequestration through Grassland Management

The livestock sector is a significant contributor to agricultural GHG emissions. Overall, emissions along the livestock supply chain, including energy use and land-use change, are estimated at 7.1 Gt CO_2-eq per year, accounting for 14.5% of all anthropogenic GHG emissions. They are mainly driven by feed production and processing as well as enteric fermentation in ruminants producing methane, representing 45% and 39% of the sector emission, respectively (Gerber et al. 2013). Thus, climate-smart management practices in livestock have a high GHG mitigation potential. The IPCC (2019) estimated the overall mitigation potential of the sector at 0.12–0.25 Gt CO_2-eq per year.

Even though most common interventions to reduce GHG emissions in livestock focus on improved production efficiency at the animal and herd level, increasing C sequestration in grasslands is also an effective tool for offsetting overall emissions. Global modelling led by the Food and Agricultural Organization (FAO) predicted potential C sequestration of various grassland management practices worldwide over 20 years (Gerber et al. 2013). They estimated a sequestration potential of 409 Mt CO_2-eq per year through improved grazing management practices in grasslands globally. Another 176 Mt CO_2-eq of sequestered emissions per year was estimated to be possible by integrating legumes in grassland areas. Hence, a combined mitigation potential of 585 Mt CO_2-eq per year could be achieved by combining these two practices, representing about 8% of overall livestock supply chain emissions. Bossio et al. (2020) estimated the combination of both practices at 300 Mt CO_2-eq per year (Figure 2-2). According to the 4th Assessment Report to the IPCC (2007), a broad range of grazing and pasture improvements on a global scale might even sequester 1.5 Gt CO_2-eq of C per year.

2.9 New Zealand – Climate Change and Agriculture

The previous sections laid the theoretical foundations regarding the role of agriculture, especially soil management, on GHG emissions and the sequestration of C in soils. As the practical work of this thesis was conducted in New Zealand (NZ), the following section will examine the specific characteristics of NZ's agricultural landscapes, particularly looking at the dairy industry. Furthermore, the countries ambitious climate mitigation and adaptation targets will be presented. Finally, a specific look is taken at the country's potential regarding SCS.

2.9.1 Farming History

To better understand current trends and developments of NZ's agriculture, it is helpful to have a brief insight into the country's agricultural history. Ecosystems of NZ have undergone profound changes over the last centuries, substantially influenced by the human settlement on the islands (McGlone 1989; MacLeod and Moller 2006). Before the arrival of the first settlers, NZ's vegetation consisted mainly of alluvial floodplain forests, species-rich wetlands, and indigenous grasslands inhabited by a variety of endemic birds and insects (MacLeod and Moller 2006).

The first Polynesians arrived around 1000 years ago on the islands, initiating far-reaching modifications to the prevailing ecosystems. Changes were most drastic around 750 and 500 years ago. Widespread forest areas were cleared with fire and used as agricultural land to provide enough food for the rapidly growing Polynesian population. Moreover, endemic animal populations like the giant Moa (a giant flightless and browsing bird) were hunted down. The over-exploitation of natural resources led to a spiral of biodiversity loss and extinction and irreversible changes to the ecological composition and landscape (McGlone 1989). However, the human-made modification was certainly not the only factor leading to environmental changes. Climatic changes, large-scale erosions, earthquakes, volcanic eruptions and naturally occurring fires played their part as well. Retrospectively, anthropogenic and natural changes are hardly distinguishable (McGlone 1989).

Another milestone in NZ's ecological modification was the arrival of the European settlers in the early 19th century, leading to an increase of exotic plants in agricultural landscapes from 35% to 60% between 1900 and the mid-1970s (Molloy 1980). The development of NZ agriculture since 1840 can be classified into five major phases: colonisation, expansion, early intensification, diversification, and later intensification (Glasby 1991; MacLeod and Moller 2006):

- During the colonisation phase (1840-1870), large areas of native grassland got burnt for grazing to increase sheep populations. However, newly established pastures were highly unproductive as nutrients quickly depleted.
- The expansion phase was driven by the introduction of refrigerated shipping (1882). Significant parts of the indigenous forest were removed and replaced by permanent pasture to meet the increasing global demand for agricultural products.
- The early phase of intensification (around 1920) was characterised by significant progress in soil sciences, fertilisers, and plant and animal breeding, which led to a substantial increase in stocking numbers and productivity, while areas of agricultural production stayed relatively stable.
- During the diversification phase (starting around the 1940s), technical developments shaped the agricultural sector. The introduction of aerial top dressing allowed the fertilisation of large infertile areas of hardly accessible hill country. The sector diversified from sheep and cattle farming to include deer, goats, horticulture, and agroforestry.
- The later phase of intensification started in the 1980s and is still ongoing. Further development of agricultural practices is leading to a steadily increasing productivity. Dairy production is rapidly growing, and the intensification of farming practices is polluting the environment.

2.9.2 Dairy Farming and its Environmental Impact

The dairy industry has been one of NZ's fastest-growing economic sectors over the last decades. However, an intensification of production processes led to extremely high and unsustainable use of external inputs like feed, water and fertilisers, resulting in severe negative impacts on the environment (environmental externalities[1]) (Foote et al. 2015). External costs, both economically and environmentally, are not paid off by the polluting farmers or industry. Instead, they have to be covered by the public, for instance, through environmental remediations paid by public taxes, health issues arising or the decline of tourism.

The environmental impact is contrasted by the high export sales of the industry. The dairy sector is the main contributor to NZ's overall export revenues, with annual revenues of NZ$17.2 billion (US$12.5 billion) (as of March 2018). Thus, it is the country's largest export sector, with a share of nearly 20% on total goods and service trading and yearly average growth of 8% per year since 1990 (NZIER 2018).

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[1] External effects, or externalities are a form of market failure. "They are said to occur when the production or consumption decisions of one agent have an impact on the utility or profit of another agent in an unintended way, and when no compensation/payment is made by the generator of the impact to the affected party" (Perman et al. 2003, p. 134).

NZ is currently the leading export country for dairy products in the world. There is a total of 1,755,148 ha of agricultural land used for dairy farming and 4,992,914 cows in the country. In contrast, stocking numbers of 1990 were still at 2,402,145 animals on an area of 1,023,545 ha. However, not only the total stocking numbers expanded. Also, the productivity increased significantly over the last decades from an average of 259 kg milk solid per cow in 1992 to 368 kg in 2018 (DairyNZ 2018).

A monetisation of environmental impacts is difficult to make. Approaches mainly estimate the costs of remediation or mitigation of ecological damage. The first nationwide assessment of external environmental costs of dairy farming in NZ estimated externalities, surpassing the 2012 overall export revenue of NZ$11.6 billion (US$8.4 billion) of the sector (Foote et al. 2015). Tait and Cullen (2006) calculated the external costs of dairy farming only in the Canterbury Region at NZ$169.59 to NZ$308.23 per ha. The highest external costs of dairy farming arise through nitrate contamination of drinking water, nutrient pollution to lakes, soil compaction and GHG emission (Tait and Cullen 2006; Foote et al. 2015).

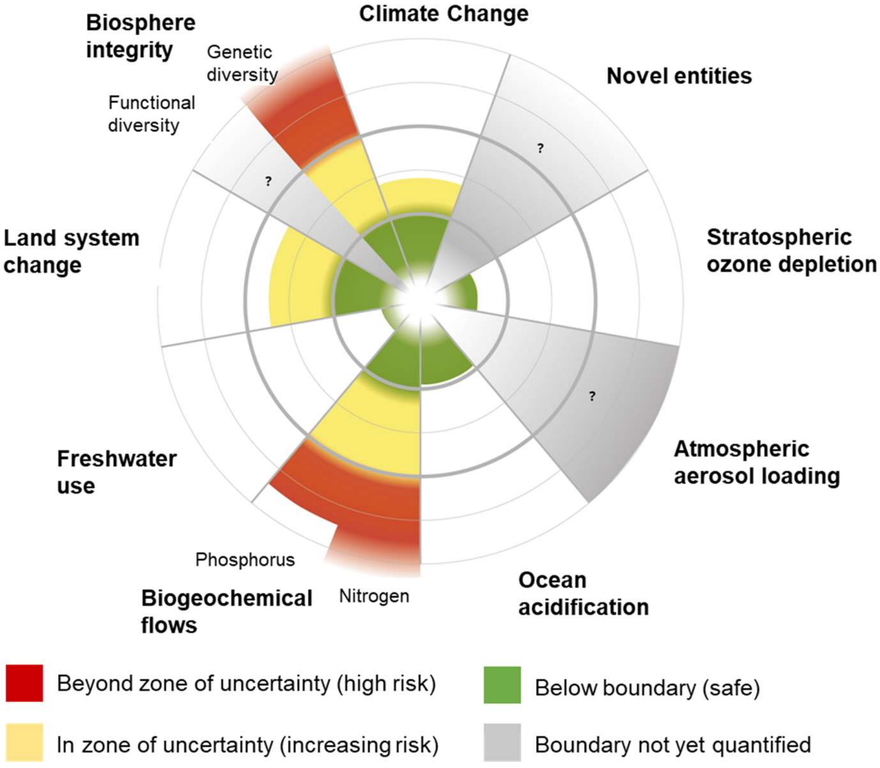

Figure 2-3: Current status of the control variables for seven of the planetary boundaries. (Steffen et al. 2015)

The planetary boundary framework provides a science-based analysis of the risk of human disturbance on the Earth system (Rockström et al. 2009; Steffen et al. 2015) (Fig 2-3). It assesses the anthropogenic influence on genetic diversity and the biogeochemical cycles of phosphorus and nitrogen as a high-risk zone of irreversible disturbance. All three are strongly influenced by intensive dairy farming.

Several concrete environmental issues arising through the intensification of dairy farming in NZ are described in more detail in the upcoming paragraphs:

Intensification of farming is connected chiefly to increased **fertiliser inputs**, compensating for nutrient losses, and maintaining high productivity. Deficits in the primary nutrients N, P and K are mainly compensated through industrial fertilisers. NZ-wide costs of fertilisers were estimated to be around NZ$593 million in 2012, and a large portion of these fertilisers is imported from politically critical countries, involving geopolitical problems (Foote et al. 2015). The high dependency of the country on fertiliser inputs becomes especially evident through the use of P-fertilizers. Fertilised P is mainly derived from mining phosphate rock, a non-renewable resource that global reserves may be depleted in around 50-100 years (Cordell et al. 2011). Sill, NZ imports around 800,000 metric tonnes of P rock annually, and its average P fertiliser consumption are three times larger than the world average and seven times larger than Europe (Li et al. 2015). The excessive use of fertiliser inputs makes the country dependent on imports and vulnerable to international market changes. At the same time, it is polluting water bodies through nutrient leaching.

The most used **brought in feed** supplement for NZ dairy cows is palm kernel expeller (PKE) (Ministry for Primary Industries 2016). PKE is a leftover from the process of oil extraction from oil palm seeds. Palm oil production is causing severe environmental damages outside of NZ. Vast areas of native rainforest are constantly cleared to meet global demands. New Zealand is the largest international importer of PKE, importing nearly one-third of global trades (Index Mundi 2020).

Dairy farming in NZ strongly impacts **freshwater quality**, mainly through bacterial contamination and nitrate leaching (Cardenas et al. 2011). Exceptionally high leaching rates have been recorded on irrigated dairy farms in Canterbury (Lilburne et al. 2010). Elevated NO_3 levels in NZ's groundwater are especially critical as about 40% of the population depends on groundwater for drinking (Rajanayaka et al. 2010). Excessive nitrogen intake can be harmful to humans, as it is linked to cardiac disease and several types of cancers (Townsend et al. 2003). Besides the negative effect on human health, excessive levels of N and P can cause algal blooms, leading to eutrophication of water bodies (Chislock and Doster 2013). A study found 44% of monitored lakes in NZ to be in a eutrophic state or worse, almost all of them near pastoral catchments (Verburg et al. 2010). The costs of water treatment in broader ecosystems can be extremely high compared to an on-farm reduction of nutrients (Foote et al. 2015).

Intensive dairy farming has several negative **impacts on soils**. Excessive use of fertilisers and other chemical soil amendments often leads to heavy metal contaminations. Furthermore, heavy machinery and high stocking rates can result in soil compaction - limiting production and increasing runoff and soil erosion (Foote et al. 2015).

In conclusion, NZ's dairy industry's actual benefits should be re-evaluated considering the environmental damage and resulting costs for society. Costs of not polluting in the first place are in almost any case far lower than costs arising through a remedy of pollution.

2.9.3 Agricultural Greenhouse Gas Emissions

NZ's gross GHG emissions increased by more than 20% since 1990, mainly due to the structural changes in agriculture. Nowadays, the agricultural sector is the largest contributor to nationwide emissions (47.8% of overall emissions) (Ministry for the Environment 2020) (Figure 2-4). This emission profile distinguishes NZ from most other developed countries, where the contribution of agriculture to gross emissions is generally low (12.2% on average) (Ministry for the Environment 2019b).

Agricultural emissions arise mainly through CH_4 production of cattle or sheep and N_2O emissions from N, deposited onto agricultural soils. Both GHG's have a significantly higher warming potential compared to CO_2 (CH_4: 38 times; N_2O: 265 times) (IPCC 2014b), making NZ the seventh-highest emitter of GHGs per person, on a global scale (16.9 t CO_2-eq per capita) (Ministry for the Environment 2019b).

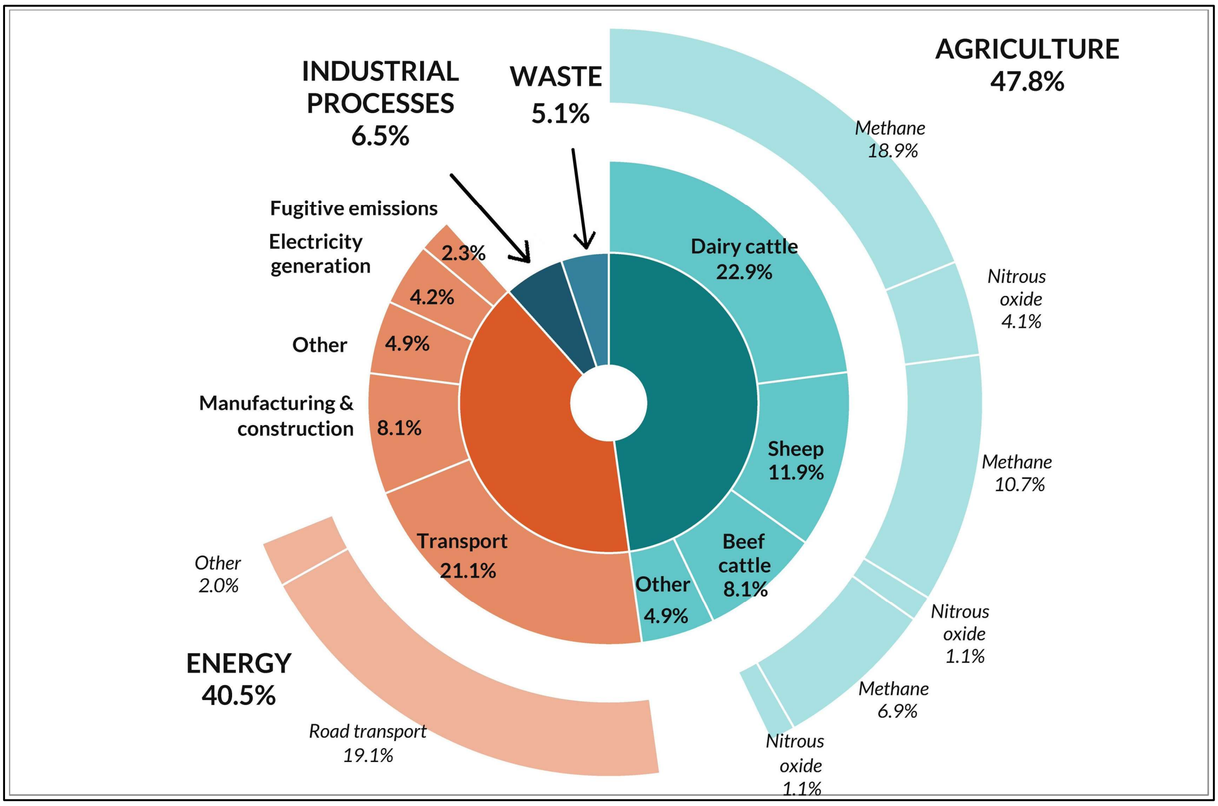

Figure 2-4: Classification of New Zealand's Greenhouse Gas Emissions in 2018. (Ministry for the Environment 2020)

2.9.4 Policy Responses to Climate Change

Climate change has shown various influences on NZ, with rising sea levels being a major threat for the islands. Over the last century, sea levels around New Zealand have risen at an average rate of 1.8 mm per year (Ministry for the Environment 2018). The IPCC predicts a global rise of the sea level by 0.2 - 0.4 m by 2060 and 0.3 – 1.0 m by 2100, depending on the emission scenario (IPCC 2014a). Most predictions would have severe impacts on the frequency of flood events in New Zealand's coastal areas. Extreme weather events such as heavy rainfall and droughts are becoming more frequent. The strongest increases of heavy rainfall events are likely to occur in the western regions and the south of the South Island. Already dry areas are likely to get even drier (Ministry for the Environment 2018). Moreover, the frequency of cyclones is increasing. In 2018, NZ was hit by two big cyclones with less than three weeks in between, destroying homes and infrastructure (Ministry for the Environment 2019a).

As a response to its comparably high GHG emissions and the arising threats for the country, NZ developed ambitious strategies to mitigate and adapt to climate change, with its agricultural sector as one central element. Furthermore, the country is highly involved in international climate policies and established a Ministry for Climate Change, responsible for all domestic and international policies regarding climate change. In November 2016, the Ministry set up the Climate Change Adaptation Technical Working Group to advise the New Zealand Government on how to adapt to the ongoing climatic changes. Consequently, the working group released their stocktake Report 'Adapting to Climate Change in New Zealand' in December 2017 (Climate Change Adaptation Technical Working Group 2017). The report focuses on adaptation while emphasising, at the same time, that adaptation and mitigation are closely linked, as an adaptation to climate change in the future will highly depend on the global level of mitigation achieved.

The Climate Change Response (Zero Carbon) Amendment Bill was set into force in November 2019 (Ministry for the Environment 2019a), defining new domestic GHG emission reduction targets for the country while setting a framework for NZ to deliver on its Paris commitment. It was the first legislation in the world to make a legally binding commitment towards the 1.5°C aim, set in the Paris Agreement, and specifies NZ's transition towards a low emission and climate-resilient economy.

The Zero Carbon Amendment Bill sets the following concrete GHG reduction targets:

- 'Reduce all greenhouse gases (except biogenic methane) to net-zero by 2050'
- 'Reduce emissions of biogenic methane within the range of 24–47 per cent below 2017 levels by 2050 including to 10 per cent below 2017 levels by 2030'

The distinction between biogenic methane and other GHGs is based on their different lifespans in the atmosphere. CO_2 is a long-lived GHG, persisting in the atmosphere for hundreds to thousands of years, while CH_4 degrades over a time span of decades. Emissions of CH_4 could therefore be at a stable rate without necessarily increasing the overall atmospheric concentration. The persistence of N_2O is between CO_2 and CH_4; in the context of the Paris Agreement, it is considered a long-lived gas (IPCC 2018).

To continuously work towards the 2050 targets, the Zero Carbon Amendment Bill establishes a series of five-year emissions budgets, specifying the quantity of emissions permitted in each period, continuously monitored and re-evaluated. To regularly re-assess the risk of climate change, National Climate Change Risk Assessments are conducted at intervals of no more than six years. The most significant threats to the country are thus identified, and a National Adaptation Plan is developed to improve the countries' resilience to climate change. To monitor the progress towards emission targets and as an independent governmental advisor, a Climate Change Commission, consisting of scientific experts with relevant expertise, was set up additionally (Ministry for the Environment 2019a).

2.9.5 Emissions Trading Scheme

The New Zealand Emissions Trading Scheme (NZ ETS) was established in 2008 as a key element of NZ's policy response to climate change. It sets a regulatory limit on emissions to participating sectors and translates them into a market price. The overall emission limit is defined by the total number of tradable emission units available on the market (Jiang et al. 2009). Every ETS market participant gets a specific number of emission units (allowance to emit GHGs) allocated by the government. Participants can decide if they want to adjust their emissions according to their number of emission units; if they want to exceed their limits by purchasing additional emission units from other participating parties; or limit their emission below the threshold and sell remaining emission units. Moreover, participants can earn additional emission units through activities that remove GHGs from the atmosphere, such as afforestation and industrial removals (Leining and Kerr 2018). While trading of emission units to and from the international Kyoto market was allowed from 2008 to 2015, NZ ETS is currently a domestic-only system.

Reducing the number of tradable units on the market allows a gradual decrease of overall emissions and a transition to a low-emission economy (Bradshaw et al. 2013). NZ ETS applies obligations to about 51% of NZ's gross emissions, almost all emissions from fossil fuels, industrial processes, and waste. It also includes unit obligations for deforestation and applies credits for eligible afforestation. Yet, it does not include unit obligations for biological emissions from agriculture (Leining and Kerr 2018).

2.9.6 Reducing Emissions through Soil Carbon Sequestration

To meet its ambitious climate goals, the New Zealand Ministry for the Environment established a nationwide Soil Carbon Monitoring System (SCMS), a statistical model estimating SOC stocks and changes in NZ's mineral soils (Lawton et al. 2002). The SCMS is based on the IPCC default methodology and combines actual soil C data, from collected samples around NZ, with spatial data sets of climate, soil type, land use and topography (similar to the methodology of Germany's GHG inventory, described in Chapter 1). Baseline soil C stocks were quantified by Scott et al. (2002) for the year 1990, stratifying the country by soil type, climate, and land use. Overall, 39 combinations of the three factors could describe 93% of the countries landscape. The baseline survey resulted in an estimate of 4192 Mt C to a depth of 1 m (Scott et al. 2002). NZ's SCMS is a combined system of direct sampling, modelling, and coefficients of change (derived from associating baseline data to historical changes in land use). It is finally creating a robust framework to estimate SOC stocks of different land-use types and the response of SOC pools to changes in land use (Lawton et al. 2002). Derived estimations are integrated into the annual reporting on the national GHG inventory, submitted to the UNFCCC (Ministry for the Environment 2019b).

Overall SOC levels in NZ are naturally at a high level. Mean stocks were recently estimated at 98.7 t C per ha to a depth of 0.3 m (Minasny et al. 2017) (Figure 2-5). The largest SOC stocks are stored in vegetated wetlands (136.06 t C per ha) (Campbell et al. 2015). Wetlands are often drained for agricultural purposes to meet the growing demand for grazing areas. Drainage favours aerobic conditions leading to oxidation of organic matter and a rapid loss of soil C, and Campbell et al. (2015) estimate the rate of C-loss through wetland drainage in NZ at 2.94 t C per ha per year. In contrast, establishing or restoring wetlands could increase nationwide SOC accumulation significantly, especially in regions unsuitable for agricultural production (Minasny et al. 2017).

Due to the overall high levels of SOC, the sequestration potential of NZ's pastures is generally low. Nevertheless, adapted management practices can prevent excessive C losses and increase C stocks, especially under intensive farming practices. These practices include avoidance of bare soil through quick sward renewals, ideally using multiple species (Whitehead et al. 2018). Furthermore, simulations by Kirschbaum et al. (2017) show a positive correlation between SOC stocks and fertiliser application rate. But since fertiliser application is already high, there is only little scope for further increase. In conclusion, practices avoiding losses and maintaining existing SOC stocks in NZ may be more rewarding than practices aiming to increase stocks.

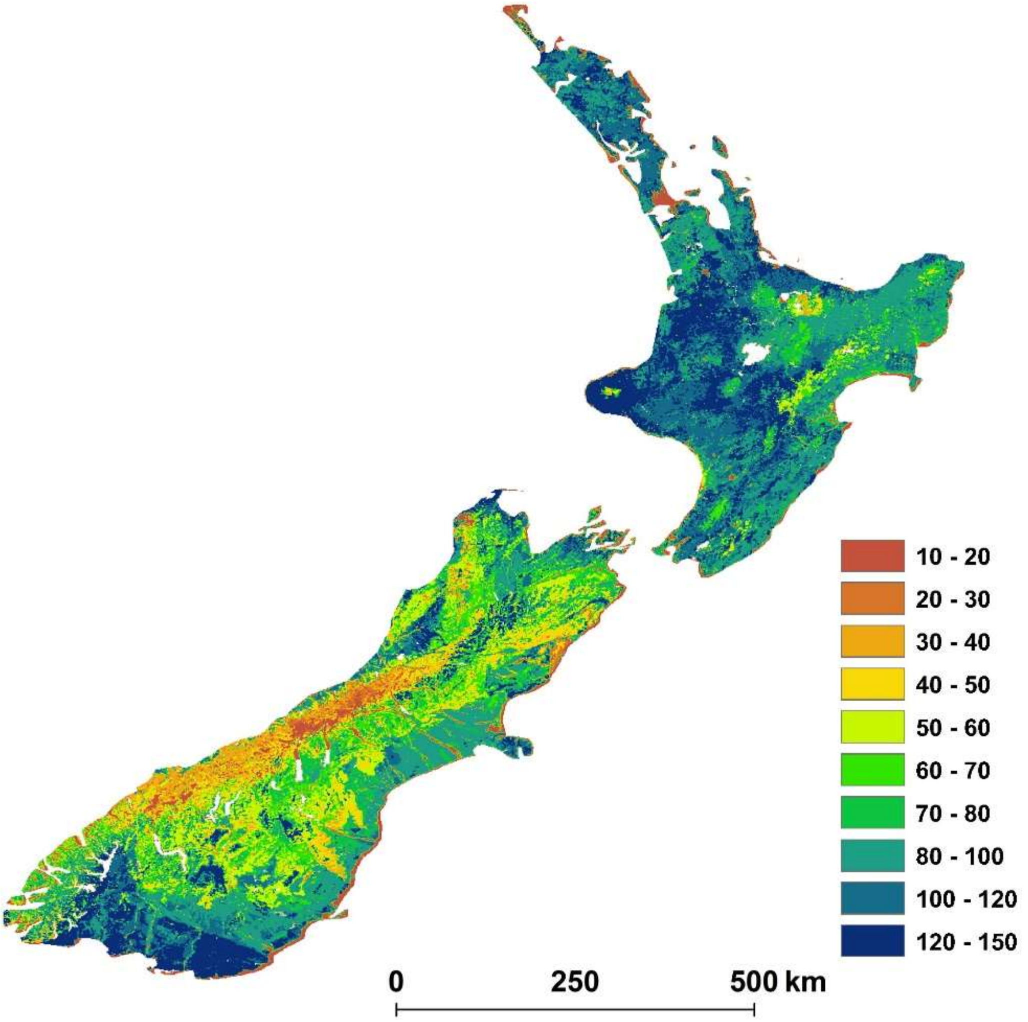

Figure 2-5: Soil C stocks of topsoil (0–0.3 m) in t C per ha of New Zealand. Data provided by the New Zealand Agricultural Greenhouse Gas Research Centre. (Minasny et al. 2017)

In the view of what has been described so far, it becomes apparent that soil management practices can be effective tools to mitigate GHG emissions while enhancing soil fertility. Adapted land use can turn soils into natural C sinks. In recent years, NZ has become a pioneer regarding its climate policies and environmental regulations. However, to achieve climate neutrality, especially in the high-emission agricultural sector, there is still a long way to go. The government must support landowners putting greater effort in C sequestration, and agricultural activities should be fully included in tradable emission markets.

Furthermore, changing trends in C dynamics have to be identified at an early stage to develop suitable strategies. C monitoring on a small, regional scale is crucial for adapted agricultural management, as broad scaled, nationwide estimates are not precise enough to show variations within single farms. To achieve all this, consistent and reliable monitoring of SOC stocks and soil gas fluxes is essential. Future research should thus focus on improving and facilitating existing methods and developing new approaches.

3 Practical Part – Case Study

3.1 Introduction and Hypotheses

The state-of-the-art review provided a thematic background regarding the climate-soil nexus and the importance of SOC sequestration for mitigating climate change and strengthening food security. The following practical section will assess how far meaningful soil gas flux measurements can be conducted on a farm level by an inexpensive and easy-to-implement approach. Therefore, a case study was conducted, using a portable chamber for in-situ flux measurements, compromising measurement series at several grassland locations within the Canterbury region, NZ. After an in-depth introduction to the methodology, the obtained results are presented, discussed, and compared to existing literature. Finally, the method's usability is evaluated under the framework conditions of the case study, and recommendations for the future use of the device in a university context are derived.

Based on literature research, the following **hypotheses** regarding the outcomes of the case study were formulated prior to the start of the measurement campaign (further described as part of the discussion):

1. The integration of trees into pastoral systems reduces soil CO_2 effluxes, and the more complex the vegetation composition, the lower the efflux.
2. Soil CO_2 effluxes are increasing with distance to tree trunks.
3. Rising soil temperatures increase soil CO_2 effluxes.
4. Soil CO_2 effluxes are highest at intermediate levels of soil moisture.
5. Soil N_2O effluxes are suppressed by trees and shrubs of the Myrtaceae family.

3.2 Material and Methods

3.2.1 Location

The presented case study is conducted at two locations, both situated within the Canterbury Low Plains ecological district, defined by the NZ Department of Conservation (McEwan 1987) (Figure 3-1). The area is located within the central east of South Island, NZ. It has undergone a 91% reduction in forest cover compared to the pre-human landscape, mainly due to the conversion of land for agricultural production (Ewers et al. 2006). Remnants of the indigenous vegetation are small and fragmented but can contain a large proportion of threatened species (Dollery 2017). The region's climate is dry with strong north-westerly foehn winds, warm summers, cool winters and low rainfall (600-800 mm), resulting in low humidity and high evaporation rates (McEwan 1987). The droughty Lismore Soils of the area are the best-known example of the stony terrace soils of the Canterbury Plains. These soils are defined by less than 45 cm of fine material (loess) overly alluvial gravels. They are free draining with a low capacity for storing water and rather infertile. Still, pasture yield can be high with irrigation, and agriculture has thus been widely converted from sheep to dairy farming in recent decades (Molloy 1993).

The first study location is 'Te Whenua Hou', an intensively used dairy farm north of the Waimakariri River (Figure 3-1). The area, referred to as 'Eyrewell forest', historically contained large areas of native forest, with approximately 6-25% kānuka (*Kunzea* sp.) forest and 1-5% kānuka-mānuka (*Leptospermum scoparium*) shrubland. Nowadays, less than 1% of the native forest remains as protected remnant areas. In the early 1930s, a large part of the area (6,764 ha) was converted into a production pine forest, consisting mainly of radiata pine (*Pinus radiata*). In 2000, the area was returned to the indigenous South Island Maori tribe (Ngāi Tahu), and the pine plantation was converted into pastoral land. As part of this, 150 ha were set aside for indigenous habitat restoration in cooperation with Lincoln University to compensate for biodiversity losses (Dollery 2017). Te Whenua Hou contains stony and free draining soils that support a mosaic of agricultural land intersected with water streams and tree-lined field boundaries.

The second study location is the 'Ashley Dene' research farm, a 355-hectare property near Lincoln University (Figure 3-1), focussing on sheep farming. At Ashley Dene, the predominance of sedimentary greywacke sandstone and siltstone gravels in the shallow stony soils forms soils low in P, Ca, Mg and Fe (typically 3% C, 0.17% N, pH 5.3). However, lime and superphosphate could increase fertility rates since the 1940s (McLenaghen and Webb 2012).

Figure 3-1: Location of the study sites within the Canterbury Low Plains (large map) and location of the Canterbury Plains within South Island, NZ (upper right). (Created with Google Earth)

3.2.2 Site Selection

Measurements at **Te Whenua Hou** were conducted at three sites (Figure 3-1 and 3-2):

- A former area of production pine forest used as an irrigated and regularly grazed pasture site for dairy cattle. In the following, referred to as **pasture**.
- A former area of a production pine forest set aside as a reserve for active indigenous habitat restoration after clearance, where seedlings (dominant species *Kunzea serotina*) were planted as part of the restoration project in 2015. In the following, referred to as **reserve**.
- A protected remnant area of indigenous native forest. In the following, referred to as **remnant.**

Figure 3-2: Measurement sites at Te Whenua Hou. a) Pasture b) Reserve c) Remnant (Photo: Ledesma)

Five spatially evenly distributed plots were selected within each site. One soil collar per plot, used as a base for the gas flux measurement chamber, was inserted into the soil and used throughout all measurements (detailed view on collars in Annex, Figure 8-2).

At the **Ashely Dene research farm**, measurements were conducted on a regularly sheep-grazed pasture site close to the farmhouse, containing scattered individual trees (*Eucalyptus nitens)*. One single standing tree (larger specimen with a height of approximately 20 m) was selected, and two transects moving away from the trunk were drawn. Soil collars were inserted into the soil within the transects, in distances of 0.5 m, 1 m, 2 m, 4 m and 8 m, respectively (Figure 3-1, 3-3 and 8-1 in Annex).

Figure 3-3: Measurement site at Ashley Dene research farm. (Photo: Ledesma)

3.2.3 Measurements of Soil Gas Fluxes

<u>Soil CO_2 effluxes</u> were measured using the LI-8100A Soil Gas Flux System (Li-Cor, Lincoln, NE, USA) (Figure 3-4). The apparatus is housing an infrared gas analyser (IRGA), connected to a portable 20 cm survey chamber, allowing direct measurements in the field. The LI-8100A was coupled to a GS1 Soil Moisture Sensor (Decagon Devices, Inc., Pullman, WA, USA), a dielectric sensor that measures volumetric water content, and an Omega Soil Temperature Probe (Li-Cor, Lincoln, NE, USA), a 2-wire thermocouple sensor.

Figure 3-4: Setup of the LI-8100A in the field (Photo: Ledesma)

Both sensors were wired to the auxiliary interface, allowing constant logging of data throughout the measures. Soil collars, made from PVC pipes (20 cm diameter), were inserted permanently into the soil. Preparation of the collars happened several days before the first measurements to prevent disruptions of the soil system. The chamber offset (extent of collars above the soil surface) was measured individually for each plot to determine the respective total system volume of the LI-8100A, used for subsequent flux calculations.

Each CO_2 measurement had an overall observation length of 120 seconds. This duration was chosen as LI-COR Biosciences (2010) recommends an observation length of 90-120 seconds at moderate to low levels of CO_2 fluxes. One observation included a 10-second pre-purge and a 20-second dead-band, where data is measured but not logged. During the pre-purge, the airflow is running while the chamber is still open, bringing CO_2 concentrations in the chamber and sampling tubes back to near ambient levels. The dead-band starts after the complete closure of the chamber to establish a continuous mixing of air within the chamber. No post-purge was used as each measurement included only one observation. The measurement cycle of the LI-8100A is illustrated in Figure 3-5, and further information can be found in the openly accessible LI-COR manual (LI-COR Biosciences 2010).

The collected data were analysed using the 'Soil Flux Pro' software (version 4.0.1), fitting an exponential function through the measured data points and calculating the CO_2 efflux in µmol m^{-2} s^{-1}.

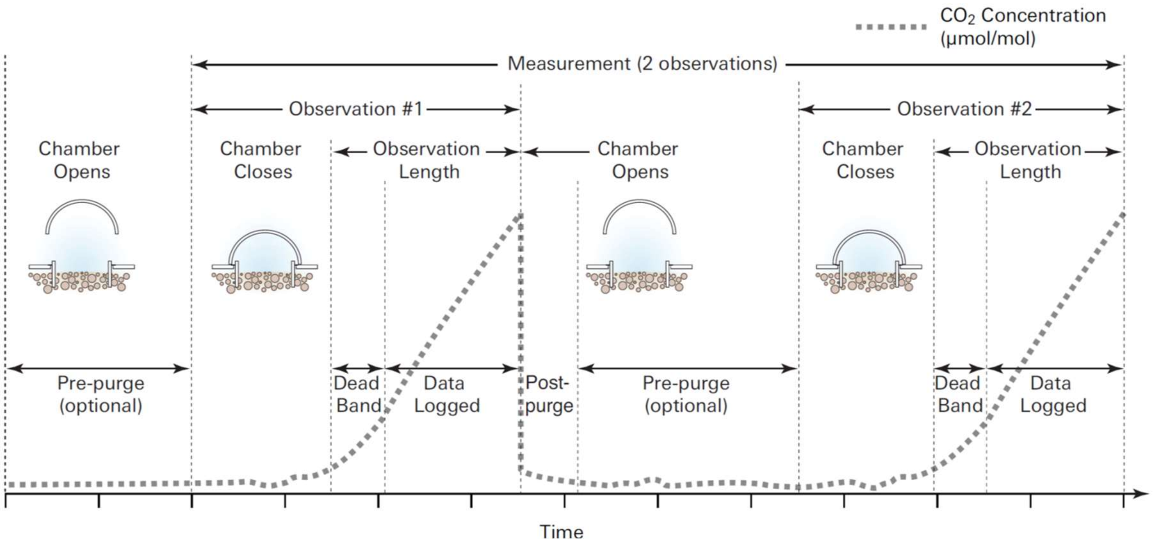

Figure 3-5: Measurement cycle of the LI-8100A. (LI-COR Biosciences 2010)

The LI-8100A is a closed dynamic chamber system (CDC), the most commonly used method in laboratory and field measurements of soil respiration (Luo and Zhou 2006, p. 162ff). CDC systems are fully enclosed during the measurement, while changes in CO_2 concentration in the system are detected. Meanwhile, the air is continuously circulating in a loop between the chamber and the CO_2-detecting IRGA sensor. Measured increases of CO_2 concentration within the system are subsequently used to estimate the efflux of CO_2 from soil to atmosphere, which can be described by the following equation (Field et al. 1989):

$$F = \frac{(c_f - c_i)V}{\Delta t A}$$

Equation 3-1: Calculation for soil CO_2 efflux

Where c_i is the initial CO_2 concentration, c_f is the final CO_2 concentration, V is the volume of the system (including chamber and tubes), Δt is the time between the measurements, and A is the soil surface area covered by the chamber. This equation shows that chamber dimensions must be known accurately, including the chamber offset. The LI-8100A is not only measuring the CO_2 start and end concentration but constantly plotting increases in concentration. Effluxes are finally estimated using the slope of a fitted linear or exponential regression, describing the concentration increase over the measurement time.

Air samples were collected using the circulating air stream of the LI-8100A to **estimate soil N_2O effluxes**. Five samples were taken manually, at evenly spaced time intervals, over a period of 30 minutes using a 10 ml syringe. The first sample used to determine ambient concentrations was collected after 20 seconds of pre-purge, allowing a uniform mixing of air within the chamber headspace prior to sampling. The collected air samples were then transferred into vacuumed vials and analysed for total N_2O concentrations using a gas chromatograph (GC, SRI 8610; SRI Instruments, CA, USA). Soil temperature and moisture were monitored in the proximity of the chamber during the measurement. The ideal gas law was then used to derive a gas efflux from the rate of increase in N_2O concentration during the measurement, as described by the following equation (Saggar et al. 2008; Saha et al. 2017):

$$F = \rho \frac{V}{A} \frac{\Delta c}{\Delta t} \frac{273}{(T + 273)}$$

Equation 3-2: Calculation for soil N_2O efflux

Where F is the gas-flux (mg m^{-2} h^{-1}); ρ is the density of the gas (mg m^{-3}); V is the volume of the chamber (m^3); A is the base area of the chamber (m^2); $\Delta c\ /\Delta t$ is the average rate of change of concentration with time (ppmv h^{-1}), and T is the temperature (°C) in the chamber. Results were finally extrapolated to g N_2O-N ha^{-1} day^{-1}. CH_4 was measured together with N_2O but not further used in flux calculations as concentrations were constantly at an undetectable level for the method used.

Soil samples were taken to a depth of 15 cm, using a soil core of 5 cm diameter. Three samples were taken within each collar after finishing the last gas flux measurement and bulked. **Total carbon and nitrogen** were analysed using a CNS Elemental Analyser (LECO Australia Pty Ltd, NSW, AU).

All measurements were conducted during the spring/ early summer months of 2019 (September – December).

3.2.4 Statistical Analysis

Statistical analysis was conducted using SPSS statistics 26 (IBM Corp. 2019). A multifactorial analysis of variance (MANOVA) was performed for both locations respectively, regarding the variables CO_2 efflux, soil temperature, soil moisture as well as C and N content; considering the factor 'vegetation type' (factor levels: pasture, reserve, remnant) or rather 'distance to tree' (factor levels: 0.5 m, 1 m, 2 m, 4 m, 8 m). Furthermore, variables were tested for linear relationships using the Pearson’s Correlation Coefficient.

3.3 Results

3.3.1 Te Whenua Hou

3.3.1.1 Soil gas fluxes

Over the whole measurement period, five measurements of CO_2 effluxes per plot were conducted at the pasture and reserve site (n=25) and four measurements at the remnant site (n=20). Two days included two measurements each to observe diurnal variations, one in the morning and one in the afternoon. The remaining days covered a single measurement in the morning.

Overall, mean effluxes were highest within the pasture site (7.38 μmol m^{-2} s^{-1}), followed by the reserve (5.1 μmol m^{-2} s^{-1}) and remnant (2.9 μmol m^{-2} s^{-1}), with descending standard deviations in the same order (3.15; 1.97; 0.82). The boxplot below illustrates the differences in CO_2 effluxes between the three sites of different vegetation (Figure 3-6). Additionally, Table 3-1 compares the means of soil moisture, soil temperature and CO_2 effluxes. It is apparent from this table that standard deviations of soil moisture are very high at all sites.

The MANOVA revealed that mean CO_2 effluxes differed significantly between sites (p=0.01 level). Moreover, mean soil moisture levels were significantly different between sites (p=0.05 level), while soil temperature was not (Table 3-1).

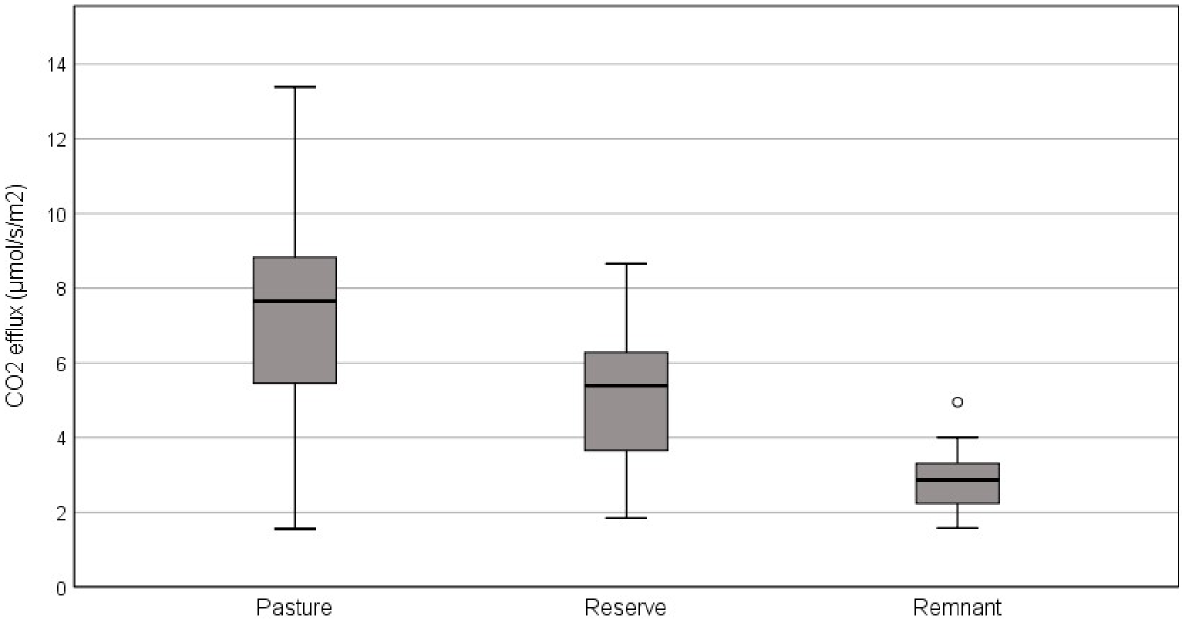

Figure 3-6: Boxplot of mean CO_2 effluxes of the three sites of different vegetation

Table 3-1: Mean values of soil moisture, soil temperature and CO_2 efflux of the three sites of different vegetation (±standard deviation). P-values indicate the level of significance of the factor 'vegetation type' regarding the respective variable. Bold values indicate significant differences ($p<0.05$).

Site /vegetation type	Soil moisture (%)	Soil temperature (°C)	CO_2 efflux (µmol m^{-2} s^{-1})
Pasture (*n=25*)	0.24 ±0.14	13.69 ±2.88	7.38 ±3.15
Reserve (*n=25*)	0.16 ±0.16	14.17 ±3.50	5.10 ±1.97
Remnant (*n=24*)	0.15 ±0.12	12.81 ±2.44	2.90 ±0.82
Total (*n=74*)	0.18 ±0.15 **(p=0.037)**	13.61 ±3.08 (p=0.34)	5.29 ±2.90 **(p=0.00)**

Including all measurements, a significant positive correlation was found between soil moisture and efflux (p=0.01 level), but there was no significant correlation between soil temperature and efflux. The data, however, indicate strong inhomogeneities in CO_2 efflux between different plots of the same vegetation, especially within pasture and reserve.

Interestingly, descriptive statistics suggest an interconnection of soil temperature and soil moisture influencing CO_2 effluxes. Low moisture levels (~<5%) seem to inhibit effluxes even though soil temperature is at a high level. However, effluxes at the remnant site seem to be less influenced by fluctuations in soil temperature and moisture, indicated by a low standard deviation of CO_2 efflux.

N_2O effluxes were measured only once per site and plot. Effluxes were generally very low or rather not measurable. Only the pasture site had overall detectable increases in N_2O concentrations after measurement intervals of 30 minutes, with an average flux of 2.34 (±0.52) g N_2O-N ha^{-1} day^{-1}. Results at the reserve site had strong spatial variabilities. Two plots did not show any increase in concentration over time. However, the other three plots revealed high variabilities (21.27; 4.27; 0.92 g N_2O-N ha^{-1} day^{-1}). At the remnant site, no detectable changes in N_2O concentration were observed at all. As measurements of N_2O had methodologic difficulties (see discussion), results were not included in statistical analysis.

3.3.1.2 Soil data

Soil samples were taken inside the collars, after the last flux measurement. Unfortunately, those samples got lost during storage at the university, and results are therefore not available. However, several additional soil samples were collected at scattered locations (at the remnant site only) and analysed for total C and N (n=11). Means of total C were 7.07% (±1.61); of total N 0.47% (±0.09), and the CN ratio was 15.08 (±1.03). The mean water content (WC) of the measured samples was 10.72%.

Additionally, soil data derived as part of a PhD work by Dollery (2017) deliver complementary information regarding characteristic soil parameters of different vegetation types in the area. The study compared topsoil of three vegetation types at Te Whenua Hou, remnants of native vegetation (same as examined as part of this thesis), grassland areas situated at the field boundaries (FB) and stands of established pine trees for timber production. Figure 3-7 presents an overview of her data. The results indicate that the remnant area has the least nutrient-rich topsoil, with the lowest NO_3^-, NH_4^+ and available P levels and the lowest pH. However, it had the highest levels of total organic C and total N. Conversely, the grassland had the highest values of pH, NO_3^-, NH_4^+ and available P.

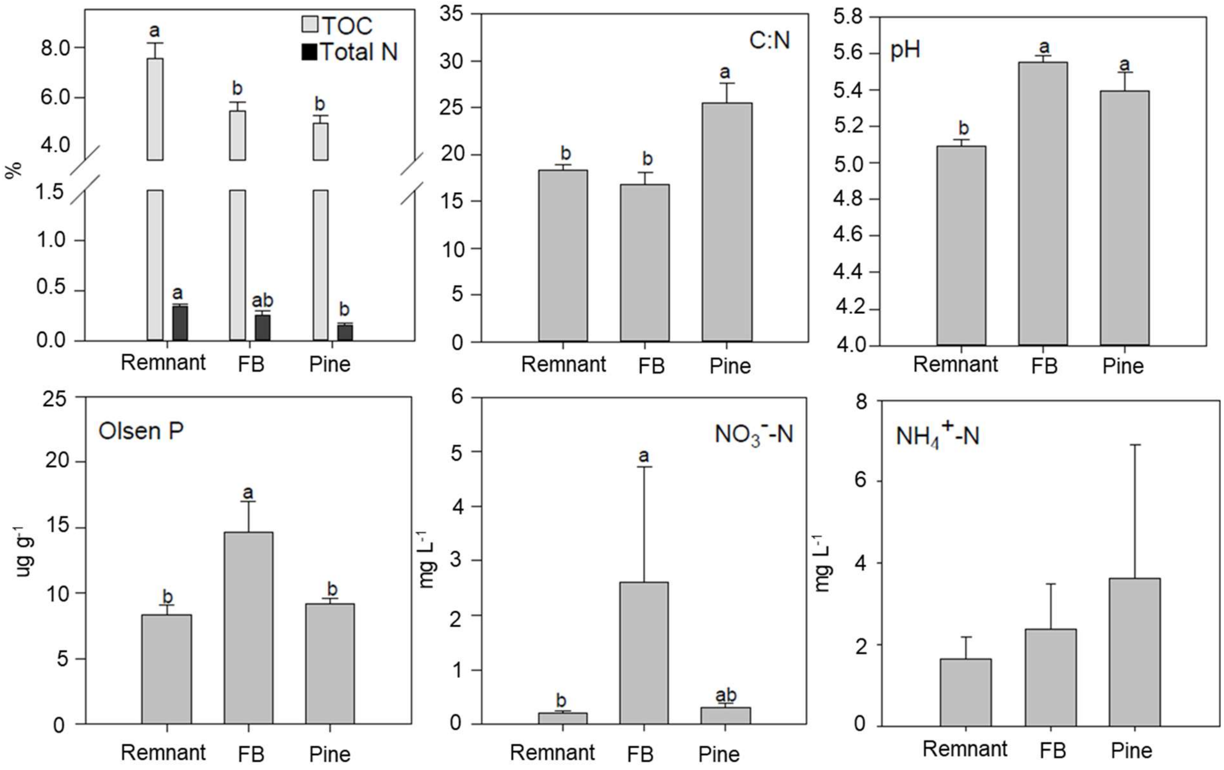

Figure 3-7: Mean values (±standard error) for soil parameters in the topsoil. 'FB' category denotes kānuka stands along field boundary. Means that do not share a letter are significantly different ($p<0.05$). No letters indicate no significant difference. (adapted from Dollery 2017)

3.3.2 Ashely Dene Research Farm

3.3.2.1 Soil gas fluxes

Measurements of CO_2 effluxes from the two transects leading away from the trunk of the Eucalyptus tree (Transect_1; Transect_2) were conducted five times in total (from mid-October until mid-November 2019; spring in NZ) in distances of 0.5 m, 1 m, 2 m, 4 m, and 8 m (n=25, per transect). Two measurement days included two measurements each (morning and afternoon). One day included only one measurement.

Figures 3-8 and 3-9 show the relationship of CO_2 efflux and C- and N-content with distance to the trunk. A striking observation is the similar trend of the two transects regarding both relationships. Effluxes generally decrease from 0.5 m to 1 m and increase from 1 m to 2 m. C- and N-contents, in turn, increase from 0.5 m to 1 m and decrease from 1 m to 2 m. However, changes between 0.5 m and 1 m are less pronounced at Transect_1. There are only marginal variations in efflux from 4 m to 8 m.

Table 3-2 and 3-3 summarise the means of CO_2 and N_2O effluxes, soil temperature and moisture, total C and N content and the associated levels of significance. Overall, there was a significant difference between distances regarding CO_2 effluxes and soil moisture. No significant difference between distances was found regarding soil temperature. Moreover, statistical analysis revealed a positive correlation between CO_2 effluxes and soil moisture (0.01 level) and a negative correlation between CO_2 effluxes and C- and N-contents (0.01 level).

N_2O effluxes at the Ashley Dene research farm were measured once for the distances 0.5 m, 1 m, 2 m, and 4 m. Surprisingly, there was no discernible trend regarding the relationship between N_2O effluxes and distance. The overall mean N_2O efflux was 8.28 (±8.43) g N_2O-N ha^{-1} day^{-1}, influenced by two outliers. Here, too, the data were not included in the statistical analysis due to methodological concerns.

3.3.2.2 Soil data

Soil samples, taken inside the collars after the last flux measurement, were analysed for total C and N (n=10). Means of total C were 7.42% (±4.01), of total N 0.53% (±0.21), and the C/N ratio was 13.51 (±1.89) (Table 3-2 and 3-3). The mean WC of the measured samples was 10.46% (±0.02). It is noticeable that samples taken at the plot closest to the tree (0.5 m) were more depleted in C and N than those at a 1 m distance (Figure 3-9).

Table 3-2: Transect_1 mean values of soil moisture, soil temperature and CO_2 efflux (±standard deviation, n=5) as well as soil C- and N-content and N_2O-N efflux (n=1). P-values indicate the level of significance of the factor 'distance' regarding the respective variable. Bold values indicate significant differences ($p<0.05$).

Distance (m)	Soil moisture (%)	Soil temperature (°C)	CO_2 efflux (µmol m^{-2} s^{-1})	C (%)	N (%)	C/N	N_2O-N (g ha^{-1} day^{-1})
0.5	12.16 ±4.71	12.16 ±0.86	7.40 ±1.48	7.29	0.49	15.0	16.53
1	7.2 ±6.31	12.43 ±1.41	5.50 ±0.91	17.5	1.05	16.7	2.82
2	16.72 ±0.97	12.67 ±0.86	8.21 ±1.44	11.3	0.70	16.1	1.04
4	13.48 ±4.11	13.39 ±1.38	9.91 ±2.01	6.30	0.46	13.7	6.4
8	23.36 ±5.04	13.67 ±1.12	9.28 ±2.02	3.57	0.34	10.5	n/a*
Level of significance	**(p= 0.002)**	(p= 0.335)	**(p= 0.012)**				

*n/a= not analysed

Table 3-3: Transect_2 mean values of soil moisture, soil temperature and CO_2 efflux (±standard deviation, n=5) as well as soil C- and N-content and N_2O-N efflux (n=1). P-values indicate the level of significance of the factor 'distance' regarding the respective variable. Bold values indicate significant differences ($p<0.05$).

Distance (m)	Soil moisture (%)	Soil temperature (°C)	CO_2 efflux (µmol m^{-2} s^{-1})	C (%)	N (%)	C/N	N_2O-N (g ha^{-1} day^{-1})
0.5	3.96 ±4.69	13.60 ±1.89	5.20 ±1.60	6.00	0.48	12.5	5.46
1	2.02 ±0.94	13.56 ±1.64	4.50 ±0.96	8.18	0.65	12.5	3.39
2	13.40 ±2.06	13.51 ±1.43	8.52 ±1.01	5.17	0.42	12.4	3.37
4	19.34 ±4.30	14.04 ±1.57	10.78 ±2.13	3.75	0.32	11.6	27.24
8	26.46 ±4.88	13.63 ±1.65	10.35 ±1.44	5.09	0.36	14.3	n/a*
Level of significance	**(p=0.000)**	p=0.991)	**(p=000)**				

*n/a= not analysed

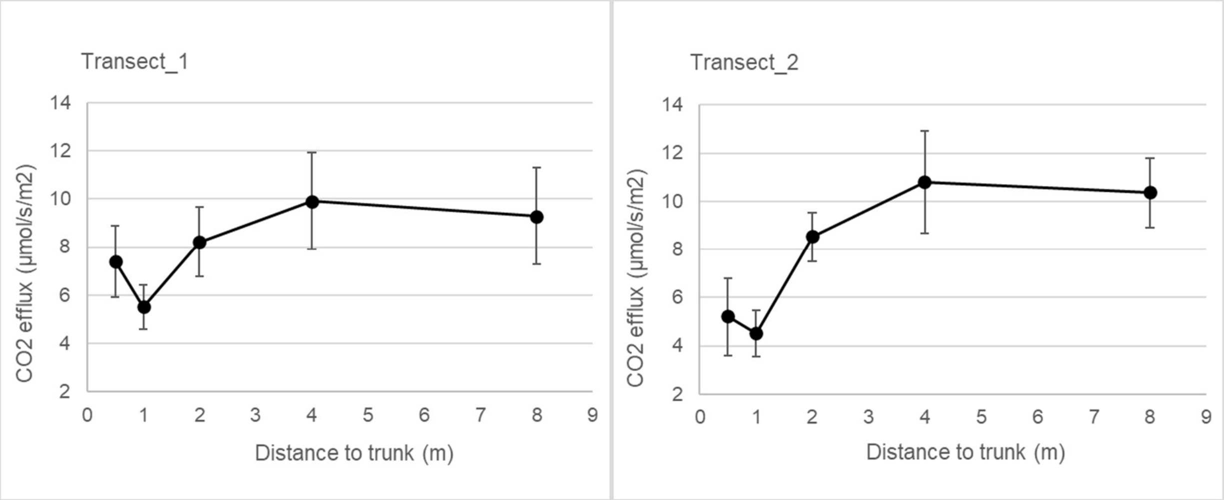

Figure 3-8: Relationship of mean CO_2 efflux with distance to trunk. Error bars indicate standard deviation (n=5).

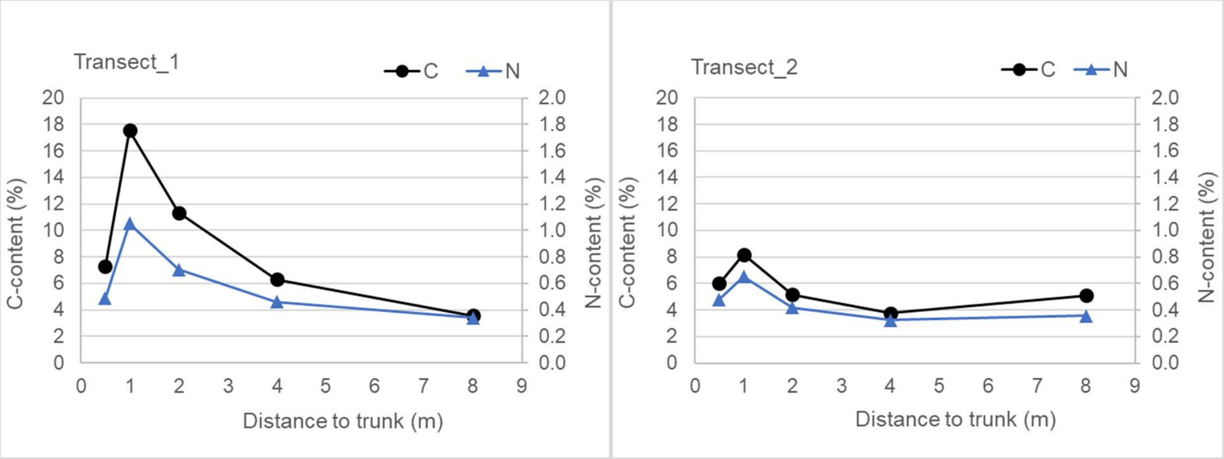

Figure 3-9: Relationship of soil C- and N-content with distance to trunk (n=1).

3.4 Discussion

3.4.1 The influence of vegetation on CO_2 fluxes

The case study results show that the composition and complexity of different vegetation types significantly influence CO_2 fluxes. A remnant of native vegetation in the Canterbury Low Plains, NZ, characterized by a high biological diversity and dominated by native species, was shown to have significantly lower soil CO_2 effluxes than a nearby irrigated pastoral area used for intensive dairy farming. Moreover, soil respiration at the remnant area was less sensitive to soil temperature and moisture changes and showed less pronounced diurnal fluctuations. In addition to that, results indicate that an active habitat restoration with a diverse composition of mainly native plants has already four years after establishment lower rates of soil respiration than the adjacent dairy pasture. The three sites compared at Te Whenua Hou, namely pasture, reserve, and remnant, had mean CO_2 effluxes of 7.38 ±3.15; 5.1 ±1.97; 2.9 ±0.82 µmol m^{-2} s^{-1}, respectively. These values lay within the broad range of soil respiration reported for temperate grasslands and forests during spring and summer months (Raich and Schlesinger 1992; Luo and Zhou 2006, p. 79ff). The case-study results will in the following be compared to literature that includes soil respiration measurements in NZ.

A study by Brown et al. (2009) investigated the regulation of soil surface respiration in a cattle-grazed pasture at the edge of the Canterbury Plains. They used a closed chamber approach to monitor GHG effluxes in the field over the course of a year. Their observed values of CO_2 effluxes ranged from 6.14 µmol m^{-2} s^{-1} (= 0.27 mg CO_2 m^{-2} s^{-1}) in late winter, with minimum soil temperatures at around 13°C, to a maximum of 10.91 µmol m^{-2} s^{-1} (= 0.48 mg CO_2 m^{-2} s^{-1}) in summer, when soil temperature reached its peak at around 20°C. The range of monitored CO_2 effluxes by Brown et al. (2009) is consistent with the case-study results, where mean effluxes of the pasture were 7.38 ±3.15 µmol m^{-2} s^{-1}. Nevertheless, a warming experiment by Graham et al. (2014) in an area of tussock grassland in central South Island found mean soil respiration levels to be significantly lower. They observed average CO_2 effluxes in an untreated control soil of 0.96 ±0.09 µmol m^{-2} s^{-1} over 27 months. A further study by Nieveen et al. (2005) used eddy covariance flux measurements to study net C exchange of grazed pastures on a drained peat soil on North Island, NZ. They found night-time respiration, which is broadly comparable to soil respiration rates under an opaque chamber, ranging from 0.4 to 8.0 µmol m^{-2} s^{-1} in winter and summer, respectively.

When comparing the results of existing literature regarding soil respiration of NZ pastoral areas, it becomes apparent that even under similar environmental conditions, soil gas effluxes might vary considerably. This inconsistency might be due to the complex interplay of biotic and abiotic factors influencing the microbial C-turnover in soils described in the literature review and further discussed in a subsequent paragraph.

Only little is found in the literature regarding soil respiration under kānuka dominated forests or shrublands. However, an extensive study by Hedley et al. (2013) compares various established stands of kānuka *(Kunzea ericoides)* and mānuka (*Leptospermum scoparium*) shrublands under different soils and climatic conditions regarding CO_2, N_2O and CH_4 effluxes over two years, using a CDC system. Under a drought-prone, stony, sedimentary soil at the south-eastern part of North Island, which is probably most comparable to the case study conditions, they calculated a mean respiration rate of 9.85 ±8.63 t CO_2-C ha^{-1} $year^{-1}$. Effluxes were ranging from values of about 5.5 µmol m^{-2} s^{-1} (= 0.24 g CO_2-C m^2 h^{-1}) to 19.9 µmol m^{-2} s^{-1} (= 0.86 g CO_2-C m^2 h^{-1}) in winter and summer, respectively. Overall, they reported strong spatial variabilities within plots of similar vegetation. Considerably lower values were observed at the kānuka stands of the case study, both at reserve and remnant. This discrepancy could be for various reasons. First, the climatic conditions between North and South Island differ substantially, shaping the temperature and water regime of the respective environment. Higher mean temperatures and precipitation of the eastern North Island are likely to cause higher CO_2 effluxes than in the case study that was conducted on the South Island. Secondly, differences in soil properties, like C- and N-content, could have influenced microbial activity. A possible explanation for the overall very low CO_2 effluxes at the remnant site might also be the thick moss layer covering large parts of the soil, retaining heat and water. Rayment and Jarvis (2000) found spatial variations in CO_2 efflux to be strongly correlated with the thickness of the moss layer in a Canadian boreal forest.

It is important to bear in mind that comparing the data derived from the case study to literature must happen with caution. Studies including soil respiration measurements are usually characterised by extensive measurements with many repetitions to account for temporal and spatial variations. Due to the restricted time frame of the thesis, results give only a small insight into the complex C-dynamics of the area. The limitations of soil respiration measurements regarding their time intensity will be explained in more depth within the final evaluation of the method.

3.4.2 Distance to trunk

Another interesting finding was that the distance to the trunk of a Eucalyptus tree on a sheep-grazed pasture significantly influenced soil respiration rates. CO_2 effluxes were generally increasing with distance to the tree. However, an initial drop in efflux from 0.5 m to 1 m could be observed in most measurements. These results are somehow contradictory to what is reported in literature. Most studies investigating the spatial variability of soil respiration at a tree-level found decreasing CO_2 effluxes with increasing distances from trunks (Wieser 2004; Epron et al. 2004; Wang et al. 2015). Wang et al. (2015) quantified CO_2 effluxes, soil temperature and moisture, and fine root biomass at transects leading away from single trunks of an apple orchard on the Loess Plateau, China, at 0.5 m and 4 m distance. They found mean soil respiration and temperature sensitivity of soil respiration (Q_{10}) to be significantly higher at 0.5 m compared to 2 m. Furthermore, they reported no significant difference in soil moisture and temperature but significantly higher fine root biomass at 0.5 m, seeming to be the main factor influencing the spatial variability of soil respiration. Similar findings were published by Wieser (2004), investigating soil respiration in a 95-year old cembran pine stand in Innsbruck, Austria, stating that spatial variation of CO_2 efflux is mainly attributable to gradients in litter mass and fine root biomass. However, the correlation of fine root biomass and soil respiration is not confirmed in other studies (Epron et al. 2004; Ngao et al. 2012). Instead, in a 3-year old Eucalyptus plantation in coastal Congo, both leaf and total aboveground litter were attributing for most of the spatial variability of soil respiration (Epron et al. 2004).

There are different possible explanations for the inconsistency of the case-study results with the presented literature. Most studies do not find soil moisture to significantly influence CO_2 effluxes at a tree-level. However, in our results, moisture levels were substantially lower in the proximity to the tree, likely inhibiting microbial respiration (Table 3-2 and 3-3). In our case, the large tree specimen was probably influencing soil moisture levels in the root zone. Moreover, we found CO_2 efflux and soil C- and N-contents to be negatively correlating. C and N generally decreased with distance, yet, an initial increase from 0.5 m and 1 m was present. The elevated levels of C and N at the proximity to the tree could be an indicator for higher litter input and fine root biomass. Both parameters were not directly measured as part of the study. Still, visual observation of the pictures in the Annex (Figure 8-1 and 8-2) indicates an accumulation of leaf litter near the trunk.

It is noteworthy that most studies chose longer distances between plots of the transects, with maximum distances of 4 m. Wieser (2004) chose distances of 0.5 m, 1.5 m and 4 m; Wang et al. (2015) only 0.5 m and 4 m. Retrospectively, fewer plots within the transects could have been established in our study, which would have saved time, allowing more repetitions instead. Finally, it is important to consider that measurements were only conducted at two transects of a single tree. Therefore, the sample is not representative, and results must be interpreted with much caution and cannot be extrapolated to a wider area.

3.4.3 Abiotic and biotic factors influencing CO_2 fluxes

Soil temperature and soil moisture are generally recognized as the main factors controlling the variability of soil respiration (Davidson and Janssens 2006). A statistically significant influence of soil moisture on CO_2 effluxes was found at all measured sites of the case study. The interconnection of soil moisture and microbial activity has been recognised for several decades, besides others by Wardle (1992), who stated that rewetting of soils could increase biological biomass. Raich and Schlesinger (1992) found highest respiration rates at intermediate levels of soil moisture. Recent findings confirm the critical role of soil moisture in decomposition processes and, consequently, soil respiration; they are presented in-depth in a global meta-analysis by Hawkes et al. (2017).

Contrary to expectations, we could not observe a statistically significant influence of soil temperature on CO_2 effluxes. This outcome is surprising insofar as temperature is reported as a major factor influencing the rate of SOM decomposition through its effect on microbial activity (Wardle 1992; Lal 2003; Davidson and Janssens 2006). However, uncertainties in the literature remain regarding variations in temperature sensitivity of different C-fractions (e.g., litter vs. humus) and pools (e.g., labile vs. stabile) (Stockmann et al. 2013). A likely explanation for the non-significant role of soil temperature in our study could be the interplay of soil moisture and soil temperature influencing soil respiration. Despite increasing temperatures, CO_2 effluxes at some plots were stagnating or even decreasing. Measurements of low CO_2 efflux and high temperatures were accompanied mainly by low soil moisture.

The interrelation of soil moisture and temperature influencing soil respiration was already described by Carlyle and Than (1988), who reported that soil respiration is not sensitive to temperature under low moisture (<7.5% volumetrically) but is more responsive to changes in temperature under high moisture levels (10 - 25%). Conversely, soil respiration is not sensitive to moisture under low temperatures (<5°C) but more responsive at high temperatures between (10° to 20° C). Harper et al. (2005) confirmed this relationship, finding soil respiration in a grassland to be more sensitive to temperature changes in relatively wet soils compared to dry soils. This relationship is well reflected in our results. Soil moisture might thus have been the main influential factor, limiting microbial activity even under elevated temperatures.

Diurnal temperature changes and variations in soil moisture were highest at the pasture site, possibly causing more pronounced fluctuations of soil respiration. A likely reason is a strong exposure to solar radiation and the relatively sparse vegetation cover at this site. In contrast, increases of temperature at the remnant site were lowest, probably because the ground is mainly covered with a thick moss layer, mediating soil temperature and moisture extremes. A combination of the moss carpet with the shade-giving kānuka trees is likely to introduce a more moderate micro-climate at the reserve site, leading to less pronounced fluctuations in temperature and moisture and lower levels of soil respiration.

Overall, the results showed strong inhomogeneities in CO_2 effluxes between nearby plots of the same site, especially at the pasture and reserve. An explanation for this might be the difference in **soil surface cover** through vegetation within the soil collars, which becomes apparent through the pictures in Annex (Figure 8-2). It has been shown that the microbial turnover of SOM is dominated by hot spots, including the rhizosphere and areas surrounding fresh detritus (Kuzyakov and Blagodatskaya 2015), being a possible explanation for the small-scale spatial variabilities between the soil collars.

Regarding the relationship of **soil C and N contents** with soil respiration, results at the Ashley Dene farm revealed a significant negative correlation. Indications for a similar trend at Te Whenua Hou can be found combining the soil data of Dollery (2017) with the flux measurements of the case study. Dollery’s results for C and N at the remnant (Figure 3-7), which are in the same range as our measured values (C: 7.07% ±1.61; N: 0.47% ±0.09), were higher compared to the grassland (significantly different for C-content). Our CO_2 effluxes were, in turn, significantly lower at the remnant compared to the pasture. It thus follows that sites of lower C and N levels have overall higher CO_2 effluxes and vice versa. This relationship was somehow unexpected, as SOC content and CO_2 effluxes are mostly found to be positively correlating (Franzluebbers et al. 2001). Nevertheless, the rate of mineralisation strongly depends on the chemical composition of the substrate (Gibson et al. 2000; Tuomi et al. 2011), being a potential explanation for our results.

The influence of N on soil respiration is diverse and very complex and can only be touched upon here. Generally, high N levels are associated with increased plant growth, affecting the rate of root respiration. Moreover, soil N influences litter decomposition and thus the rate of microbial respiration (Luo and Zhou 2006, p. 99f)

The significantly lower CO_2 effluxes at the remnant site, despite higher C and N levels, might indicate a higher recalcitrance of the SOC compared to the pasture site, suggesting an elevated potential of long-term C-sequestration under kānuka trees. Nevertheless, more extensive soil sampling and quantification of C pools through SOM fractioning would be needed to confirm this assumption.

In most cases, soil respiration is affected by a complex interplay of multiple factors, hardly separable. Due to this complexity, mechanistic and quantitative projections on how multiple environmental factors will influence soil respiration are yet not possible (IPCC 2019). Like most physiological processes of plants and microorganisms, soil respiration is finally determined by the most limiting factor (Luo and Zhou 2006, p. 104). Presumably, the most limiting factor in our study was soil moisture, as it was generally at a low level. At those measurements where soil moisture was at an intermediate level, an interplay of temperature and moisture probably accounted for most variabilities in soil respiration. Litter input and the decomposability of SOM is likely to have also caused some of the variability. However, further research taking these factors through additional experiments into account will need to be undertaken.

3.4.4 N_2O fluxes

Measured N_2O effluxes were overall at a very low level, although several outliers influenced the results. At Te Whenua Hou, only the pasture had overall detectable levels with an average flux of 2.34 (±0.52) g N_2O-N ha^{-1} day^{-1}. Only three out of five plots had detectable N_2O effluxes at the reserve, characterised by high variabilities. No detectable levels were found at the remnant. At Ashley Dene, mean N_2O effluxes were at 8.28 (±8.43) g N_2O-N ha^{-1} day^{-1}, also influenced by outliers. No noticeable influence of the proximity to the tree on N_2O fluxes could be observed.

First, it should be said that the N_2O data of the case study must be interpreted with much caution. As measurements were time-intensive (measurement period of 30 minutes per plot), N_2O effluxes were measured only once; results might thus not be transferable due to the small sample size. Moreover, N_2O concentrations in the sampled vials were often at a low level close to ambient concentrations. Air samples (10 ml each) were taken at single points in time (every 7.5 minutes) out of the circulating air stream of the closed chamber system (including chamber and tubes). As the air within the entire system was probably not perfectly evenly mixed, the air composition in the samples might not have been accurately representing the whole chamber system. Due to the overall low concentrations of N_2O in the samples, results are very prone to errors. This methodological problem became especially evident in measurements, where N_2O concentrations in the chamber system decreased in the run of a measurement period, which should theoretically not be possible.

Similar to soil respiration, the microbial formation of N_2O is strongly influenced by soil temperature and moisture. Soil moisture is particularly influential as it determines oxygen availability and soil redox potential (Butterbach-Bahl et al. 2013). Furthermore, the amount of N inputs (e.g. fertilizer, amendments) and N-outputs (e.g. grazing, harvesting), as well as the predominant vegetation, are also greatly influencing microbial processes involved in the formation of N_2O (Schaufler et al. 2010) (described in more depth in the literature review).

A meta-analysis by Saggar et al. (2008) collected and compared soil N_2O flux data from several published and unpublished studies, comparing various land-use and land-management systems in New Zealand. They found the highest N_2O emissions in dairy-grazed pastures (10–12 kg N_2O–N ha^{-1} $year^{-1}$), followed by sheep-grazed pastures (4–6 kg N_2O–N ha^{-1} $year^{-1}$) and lowest emissions in forest, shrubland and ungrazed pastures (1–2 kg N_2O–N ha^{-1} $year^{-1}$). They also identified spatial variability of CH_4 and N_2O fluxes from grazed pastures as the largest source of uncertainty when trying to obtain representative data. Compared to the findings of Saggar et al. (2008), the measured N_2O results of the pasture at Te Whenua Hou of 0.85 ± 0.19 kg N_2O–N ha^{-1} $year^{-1}$ (= 2.34 (±0.52) g N_2O-N ha^{-1} day^{-1}), lay within the lower range of ungrazed pastures. Same as for soil respiration, low moisture levels probably exerted a strong influence on microbial N_2O production. At Ashley Dene, mean effluxes were 3.02 ±3.08 kg

N_2O–N ha^{-1} year^{-1} (= 8.28 (±8.43) g N_2O-N ha^{-1} day^{-1}), laying within the range of sheep-grazed pasture, and thus representing a realistic value. Same as reported by Saggar, the spatial variability of grazed pasture was a large source of uncertainty in our case study. Outliers were likely caused by N-inputs through animal excrements within the collars prior to measurements, resulting in increased microbial activity and N_2O emissions. Findings by Hedley et al. (2013) confirm the observed low natural background levels of soil N_2O emissions from kānuka shrublands. They reported emission ranging between negligible and 0.30 ±0.20 kg N_2O–N ha^{-1} year^{-1}.

Eucalyptus and kānuka belong to the Myrtaceae family, which has recently been shown to produce antimicrobial compounds that might inhibit microbes involved in biological nitrification and denitrification. Thus, Myrtaceae trees could suppress the microbial production of N_2O, which can be especially severe after the application of N-fertilizers (Franklin et al. 2017). Whether or not inhibition of nitrification through Myrtaceae influenced the generally low levels of N_2O cannot be clearly answered through our study. More research on this topic needs to be set out before the association between Myrtaceae and N_2O production can be more clearly understood.

3.5 Evaluation of the Method

The use of soil chambers is generally seen as the most straightforward approach to measure soil gas effluxes (FAO 2019). Nevertheless, the method has its weaknesses and uncertainties, especially regarding its small spatial representativeness (point-in-space measurements only), poor temporal resolution and problematic use in long-term studies, where chambers might alter soil properties in the long run (Smith et al. 2010). However, technical improvements of instrumentation and developments in data acquisition and processing over the last decades were able to considerably improve the reliability of measurements (Smith et al. 2020).

Soil CO_2 effluxes are generally determined by strong temporal variations concerning various time scales. They range from short-term, diurnal and weekly variations to seasonal, interannual or even longer time scales (Luo and Zhou 2006, p. 108ff). Scientific studies using soil chambers to evaluate soil gas effluxes are usually characterized by high amounts of repetitions over extended periods, typically covering a whole year or even longer, to compensate for temporal variations. The annual and interannual variations of soil respiration are well illustrated by Figure 3-10.

Seasonal variations were found to occur in almost all ecosystems, while soil respiration is usually highest in summer and lowest in winter. However, soil moisture is often a limiting factor inhibiting microbial activity over dry summer months (Hawkes et al. 2017).

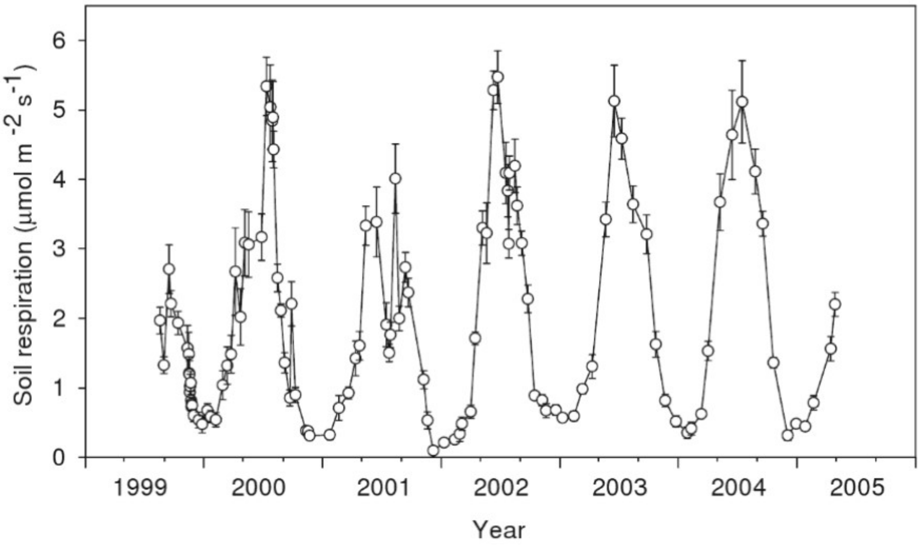

Figure 3-8: Measured rate of CO_2 efflux in a tallgrass prairie of Oklahoma, USA from 1999 to 2005. Circles represent data points and bars indicate standard error. (Luo and Zhou 2006, p. 7)

Diurnal variations can be a major source of error in interpreting results when not appropriately taken into account. They are mainly caused by the close linkage of soil respiration to changes in soil temperature over the course of the day (Xu and Qi 2001). For that reason, most studies estimating soil respiration conduct measurements at regular time intervals within one day, starting at night or early morning and finishing in the late afternoon (FAO 2019). However, Xu and Qi (2001) found that measurements taken between 09 am and 11 am tend to be most representative to estimate mean daily fluxes. Short-term variations in soil respiration, in timescales of seconds to minutes, can be caused by disturbances of the soil system before or during the measurement. Such disturbance could, for instance, be pressure exerted on the adjacent soil, resulting in a sudden increase of CO_2 outgassing. Short-term variations can be accounted for by performing multiple consecutive measurements cycle, reducing the measurements error (compare to Figure 3-5).

The above-explained shows that extensive measurements with many repetitions are essential to compensate for temporal fluctuations and derive reliable data on soil gas effluxes. However, the presented case study was often not able to meet these demands. One reason was certainly the limited workforce, as most measurements were conducted by a single person, which restricted the extent of the study. Another reason was the location and distribution of the measurement sites. Most sites were located at a two-hour drive from the university, which made measurements starting at night or early morning complicated. Moreover, sites were not always contiguous, resulting in regular breaks between the measurements, making it hard to derive meaningful comparisons between them. Finally, the scope of the case study had to be adapted to the overall timeframe of the master's thesis. Thus, measurements could only be conducted over three months, including a major part of the experimental design and the familiarization with the methodology. Nevertheless, literature shows that both the season (spring/ early summer) and the time of the measurement (late morning/ early afternoon) chosen for our study are generally well suited for measurements of soil gas effluxes.

Another primary source of uncertainty in soil gas flux measurements is the spatial variability of soil respiration, depending on various parameters such as soil properties, biological conditions, and nutrient availability. Substantial spatial variability already occurs at scales of centimeters, where variations in soil aggregation and microbial activity can create 'hot spots' of CO_2 flux on a micro-scale (Stoyan et al. 2000). While variations on a centimeter-scale can be compensated with an appropriate chamber design, variations on a meter-scale or larger must be dealt with a suitable experimental design (e.g. replicates, numbers and distribution of measurement points). Adapted sampling techniques, such as random sampling and stratified sampling with adequate replicates, can account for such spatial variability. Rodeghiero and Cescatti (2007) found that a stratified sampling method can considerably reduce the uncertainty compared to random sampling. In grasslands, they found stratified sampling to reduce the error of the annual CO_2 efflux by 12%.

Spatial variability of CO_2 effluxes has been a significant source of uncertainty in the case study, which became evident through strong variations within the same vegetation site. I identified the main influential factors on spatial variability to be the uneven distribution of vegetation cover and variations in soil properties, as described in the discussion. In addition, it should be mentioned that the soil, especially at Te Whenua Hou, was very stony, leading to substantial spatial variations in physical soil properties, thus influencing the gas flow of CO_2 in the porous soil medium. It is complicated to reduce spatial uncertainties in a study of a small scope. The establishment of more sampling points with a stratified distribution could have reduced the uncertainty but would have been connected to considerably more work.

In conclusion, accurate measurement of CO_2 fluxes is very challenging as soil respiration is highly heterogeneous in time and space. Accounting for temporal and spatial variabilities requires sophisticated experimental designs and extensive measurements. However, even for research on a smaller scale, careful planning can considerably reduce the uncertainties of results.

As described in the discussion, the case study revealed methodological issues regarding the measurements of N_2O using the Li-Cor field apparatus under the prevailing framework conditions. The method has been shown to be especially prone to errors when used on soils with low natural levels of microbial N_2O production. Possibly, extended measurement periods with longer time intervals between the air sampling could have delivered more reliable results, as N_2O concentrations within the chamber system were only slowly increasing. However, prolonged measurement periods would make the determination of N_2O effluxes even more time intensive.

3.6 Conclusions

The initial objective of the thesis was to evaluate the usability of in-situ soil gas flux measurements on a farm-scale using a portable CDC system. A comprehensible approach was needed that could be easily implemented into broader restoration projects, scientific studies or agricultural monitoring. Therefore, we set out a case study focussing on the influence of vegetation complexity and the implementation of trees on soil gas effluxes in NZ's pastoral systems. Overall, the study was able to identify high variabilities of CO_2 effluxes at a small scale and quantified factors influencing C and N dynamics under different vegetations. Interactions of these factors are highly complex, and the discussion could only emphasise some of the possible interplays.

Returning to the hypothesis posed at the beginning of this section, it is now possible to state that vegetation composition significantly influenced soil respiration in our case study and that CO_2 effluxes were lowest under remnants of native vegetation. Results also suggest that the proximity to trees affected soil respiration. Regarding the effect of soil temperature and moisture, we found soil moisture to be the most influencing parameter under the prevalent, primarily dry, conditions, limiting soil respiration even under high temperatures. N_2O effluxes were all over at a low level, but whether this was influenced by nitrogen inhibition through Myrtaceae can so far not be answered.

Compared to the usual extensive research investigating soil respiration, the relatively small scope of the case study strongly limits the scientific reliability and generalisability of our findings. Results must thus be treated with much caution and might not be transferable to a wider area. Notwithstanding these limitations, the study was indeed able to meet its primary objective, evaluating the feasibility of the method in the field.

After a short familiarisation phase, measurements of CO_2 effluxes with the LI-8100A are easy to conduct. The enclosed manual of the device gives thereby a good insight into relevant functions and settings. Derived data were generally within the range of what is reported in the scientific literature for measurements under similar conditions. More complicated, however, is the planning of a well-thought and sophisticated experimental design. A good experimental design can significantly reduce the overall measurement error. It is thus paramount to invest sufficient time into the planning phase. Stratified sampling and focussing measurements on the morning hours are only a selection of possible recommendations improving the reliability of measurements (Xu and Qi 2001; Rodeghiero and Cescatti 2008).

An arguable weakness of the case study was that each measurement consisted of a single observation only. The use of multiple consecutive measurement cycles could have reduced the short-term variations of CO_2 effluxes. This possibility should be considered in future studies.

Special attention in further trials must also be paid to the adequate use of moisture and temperature probes. Both sensors should be inserted to similar depth and as close as possible to the soil collars to derive meaningful results. Furthermore, it is advisable to place the sensors at the same spot during all repetitions. Especially at Te Whenua Hou, the stony soil made it difficult to correctly insert the sensors, which might have influenced the results. Grazing animals are another common source of error. Excrements of sheep or cattle on the measurement plots can strongly influence the microbial turnover of C and N. It is thus advisable to exclude animals from the study sites at least several days before measurements.

When taking the above-explained into account and investing sufficient time into a sophisticated experimental design, I would finally recommend implementing the LI-8100A to measure CO_2 effluxes in research activities and agricultural monitoring, even if expertise is limited. Derived data can deliver valuable indications about C dynamics in the investigated area.

The case study revealed several methodological issues regarding the measurements of N_2O using the LI-8100A under the present environmental conditions. The method was shown to be especially prone to errors when used on soils with low natural levels of microbial N_2O production. Possibly, longer measurement periods with more extended time intervals between the air sampling could have produced more reliable results, as N_2O concentrations within the chamber system were only slowly increasing. However, prolonged measurement periods would make the method even more time intensive. As the aim of the thesis was to identify an easily implementable method, I would finally not recommend N_2O measurements under the environmental conditions of the study.

In order to build on the case study's results, future research could focus more on the analysis of soil chemical and physical properties of the investigated sites. Linking soil properties to the derived flux measurements would significantly increase the conclusiveness of the results. Moreover, an extended measurement period would allow the investigation of seasonal fluctuations. Our case study only focussed on soil respiration, neglecting the photosynthetic offset of the vegetation. The photosynthetic uptake of CO_2 must be included to get a complete picture of the different vegetation types' net carbon balances, which could be done through additional flux measurements using transparent chambers.

The overall reliability and repeatability of small-scale measurements of CO_2 fluxes must still be significantly improved to be included in national GHG inventories and C markets. Therefore, a key policy priority should be the development of more comprehensive protocols facilitating suitable studies in the future.

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7 Bibliography

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8 Annex

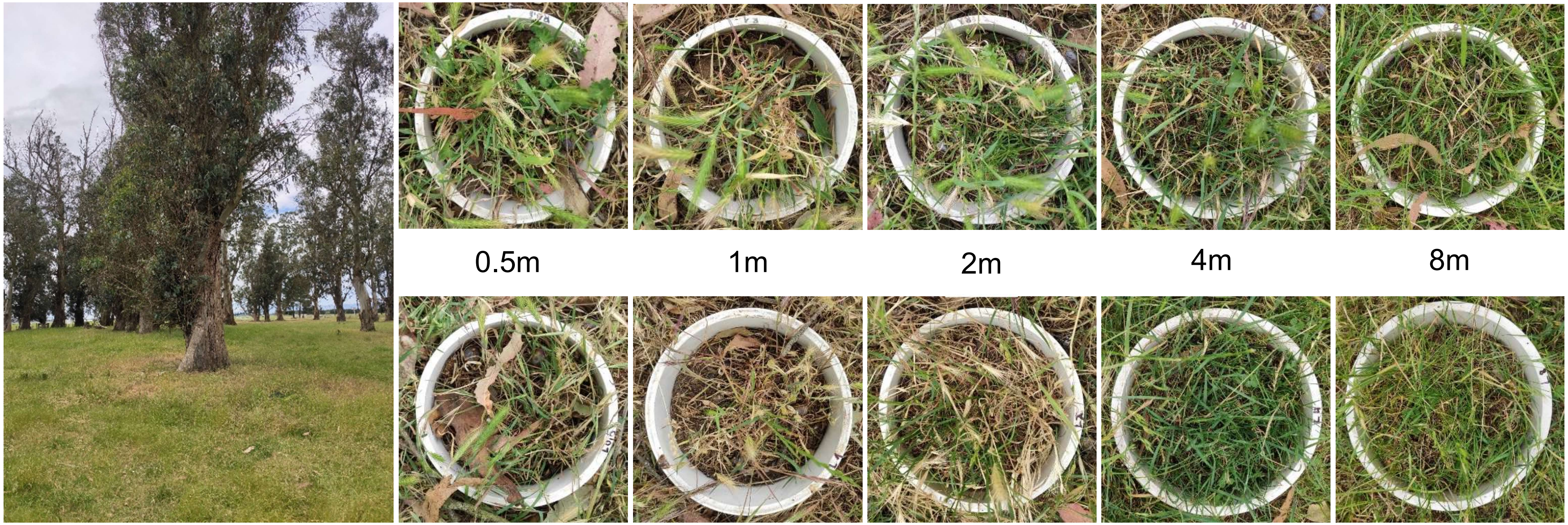

Figure 8-1: Detailed view on the soil collars used at **Asheley Dene** farm. Left picture shows investigated Eucalyptus specimen. Upper row shows collars of **Transect_1**. Lower row shows collars of **Transect_2.** (Photo: Ledesma)

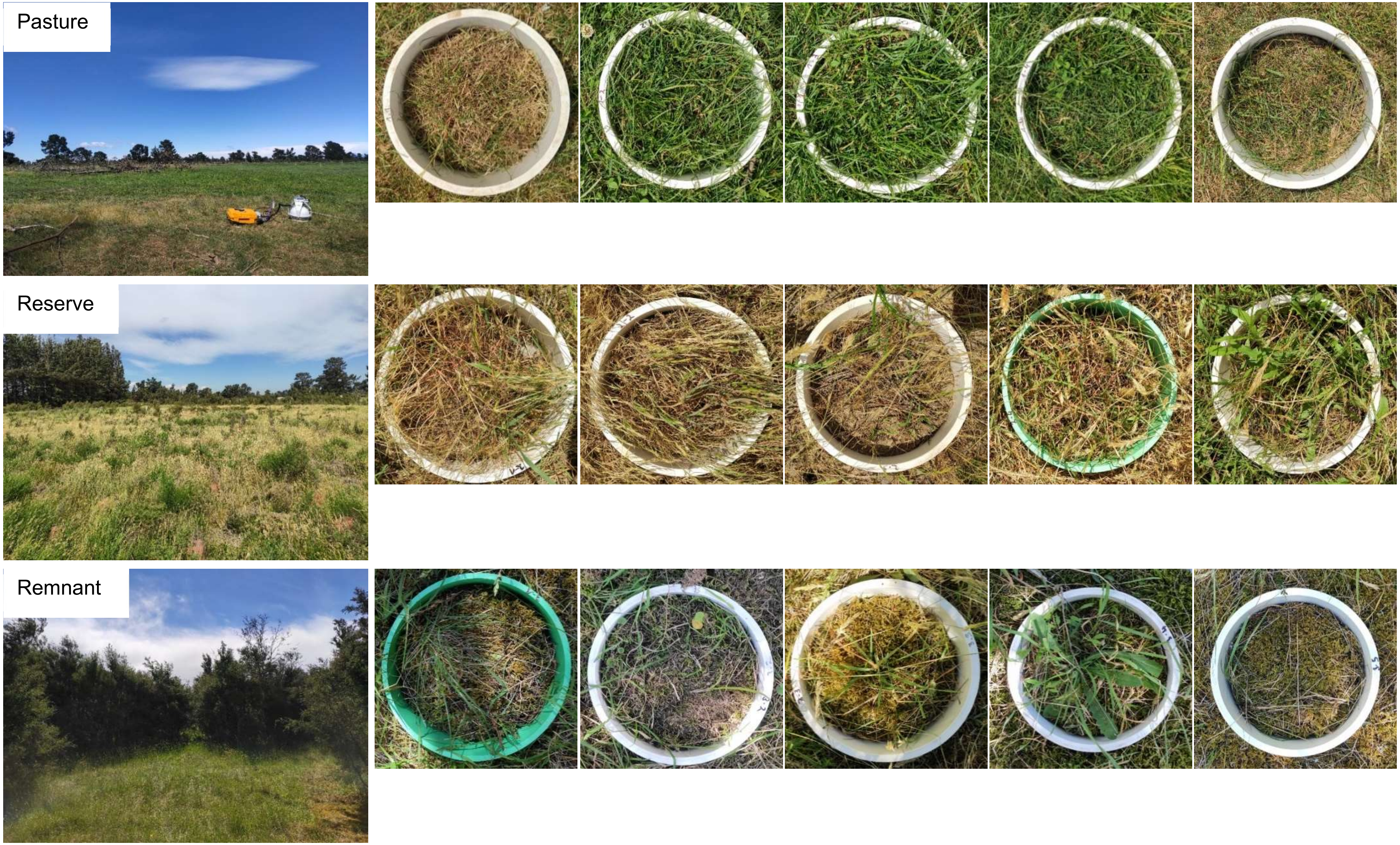

Figure 8-2: Detailed view on the soil collars used at **Te Whenua Hou** for **pasture**, **reserve**, and **remnant**. (Photo: Ledesma)