Comparison of greenhouse gas mitigation costs in cropping systems: case studies from USA, Brazil, and Germany

D I S S E R T A T I O N

zur Erlangung des akademischen Grades
Doctor rerum agriculturarum (Dr. rer. agr.)

eingereicht an der
Lebenswissenschaftlichen Fakultät der Humboldt-Universität zu Berlin

von
M. Sc. Daniel Felipe Tudela Staub

Präsidentin der Humboldt-Universität zu Berlin

Prof. Dr. Julia von Blumenthal

Dekan der Lebenswissenschaftlichen Fakultät der Humboldt-Universität zu Berlin

Prof. Dr. Dr. Christian Ulrichs

Gutachter/innen

1. Prof. Dr. Annette Prochnow
2. Prof. Dr. Jochen Kantelhardt
3. Prof. Dr. Folkhard Isermeyer

Tag der mündlichen Prüfung: 17.11.2023

Erklärung

Hiermit erkläre ich, die Dissertation selbstständig und nur unter Verwendung der angegebenen Hilfen und Hilfsmittel angefertigt zu haben.

Declaration

I hereby declare that I completed the doctoral thesis independently based on the stated resources and aids.

Braunschweig,

Daniel Felipe Tudela Staub

Abstract

Greenhouse gas emissions from crop productions systems represents a considerable share of the total man-made emissions. Nitrogen fertilization and soil management are the main sources of these gases from these systems. The fertilization and soil management strategies from the farms depend on the local conditions in which they operate, such as climate, soil, and availability of inputs, among others. Correspondingly, the feasibility of adopting mitigation strategies and their resulting effects and implications also depend on these aspects. These can be expected to significantly vary across farms in different regions. Considering the need to take quick action against climate change, it is necessary to understand which mitigation strategies profile as the most cost-efficient. Against this background, this thesis analyses mitigation strategies designed for cropping systems to evaluate their potential and costs and compares them. Case studies for selected regions in the USA, Brazil and in Germany were conducted, assessing one crop in each region. A combination of scientific literature and focus groups with local consultants and growers were used to generate realistic results which depict the local context.

This thesis identified that four strategies are feasible in the USA, three in Brazil, and only two in Germany. Some of the strategies evaluated in the farms promote the sequestration of carbon in the soil. This potential is limited in time and reversible, indicating that the strategies must be permanently used by the farms, although no additional sequestration occurs after the potential is realized. Consequently, the economics of the mitigation strategies depended on the time horizon considered. Overall, assuming a short term, strategies promoting carbon sequestration were indicated to offer a larger potential than strategies only abating nitrogen-based emissions. However, in the long term, this distinction was no longer valid.

In the short term, the lowest mitigation costs were attained by optimizing the nitrogen rate. This strategy was feasible in the USA and Brazil. These costs are indicated to be negative, implying that the growers adopting the strategy would not only lower emissions, but also reduce their costs. This strategy in Germany has among the highest costs from all the strategies studied. The next most cost-efficient strategies were revealed to be the reduction of the tillage intensity in the USA and adoption of cover crops, which was feasible in the three cases with comparable mitigation costs. However, uncertainties regarding the estimation of the carbon sequestration potential resulted in a large variation between the best and worst scenarios, with the exception of Brazil. Lastly, the adoption of enhanced-efficiency fertilizers (inhibitors), feasible in the USA and in Brazil, had among the highest mitigation costs. Assuming the long term, the mitigation costs of strategies promoting carbon sequestration increased, becoming similar to or higher than strategies without this potential, except for the reduction of the tillage intensity, which has a lower cost in this time perspective.

Additionally, the possibility of combining all the feasible mitigation strategies in each region was also evaluated. This combination maximized the mitigation potential achievable in each case. The highest potential in the short term could be attained in the USA and was approximately four times higher than in the other cases. Nonetheless, the costs were similar in the USA and in Brazil, but higher in Germany. In the long term the mitigation potentials in the USA and in Brazil are similar, but costs in Brazil were the lowest, followed by the USA and lastly Germany.

Overall, this thesis analyzed the implications of adopting mitigation strategies in cropping systems in a realistic context to derive comprehensive results. By applying the same methodology in each case, the results are comparable and can be expanded in the future by replicating the approach to include additional regions and crops. Additionally, while the findings are specific to the context in which they were calculated, they can provide insights for similar regions.

Kurzfassung

Treibhausgasemissionen aus Ackerbausystemen bilden einen erheblichen Anteil an den gesamten anthropogenen Emissionen. Stickstoffdüngung und Bodenbearbeitung sind die Hauptquellen dieser Emissionen. Die Dünge- und Bodenbearbeitungsstrategien der landwirtschaftlichen Betriebe hängen unter anderem von den regionalen Bedingungen ab, wie Klima, Boden und Verfügbarkeit von Betriebsmitteln. Dementsprechend sind auch die Machbarkeit von Minderungsstrategien und die daraus resultierenden Auswirkungen und Folgen dieser Aspekte beeinflusst. Es kann davon ausgegangen werden, dass die Bedingungen zwischen den Betrieben in verschiedenen Regionen erheblich variieren. Angesichts der Notwendigkeit, schnell Maßnahmen gegen den Klimawandel zu entwickeln, ist es erforderlich zu verstehen, welche Minderungsstrategien die kosteneffizientesten sind. Vor diesem Hintergrund analysiert diese Arbeit Minderungsstrategien für Anbausysteme, um deren Potenzial und Kosten zu berechnen und zu vergleichen. Fallstudien wurden für ausgewählte Regionen in den USA, Brasilien und in Deutschland durchgeführt, wobei jeweils eine Anbaukultur ausgewählt wurde. Eine Kombination aus wissenschaftlicher Literatur und Fokusgruppen mit lokalen Beratern und Landwirten wurde eingesetzt, um realistische Ergebnisse zu erzielen, die den lokalen Kontext widerspiegeln.

Diese Arbeit zeigt, dass vier Minderungsstrategien in den USA, drei in Brasilien und nur zwei in Deutschland umsetzbar sind. Einige der untersuchten Strategien fördern die Kohlenstoffbindung in den Böden. Dieses Potenzial ist zeitlich begrenzt und reversibel, was bedeutet, dass diese Strategien von den Betrieben dauerhaft angewendet werden müssen, obwohl nach der Realisierung des Potenzials keine zusätzliche Bindung erfolgt. Dementsprechend ist die Wirtschaftlichkeit der Minderungsstrategien von dem betrachteten Zeithorizont abhängig. Insgesamt wurde festgestellt, dass Strategien, die die Kohlenstoffbindung fördern, bei einer kurzfristigen Betrachtung ein größeres Potenzial bieten als Strategien, die nur stickstoffbasierte Emissionen verringern. Bei der langfristigen Betrachtung war diese Unterscheidung jedoch nicht mehr gültig.

In der kurzfristigen Betrachtung wurden die geringsten Minderungskosten durch die Optimierung der Stickstoffmenge erreicht. Diese Strategie war in den USA und Brasilien umsetzbar. Die Kosten sind negativ, so dass die Betriebe mit dieser Maßnahme, nicht nur ihre Emissionen, sondern auch ihre Kosten senken würden. In Deutschland liegen die Kosten für diese Strategie unter den höchsten von allen untersuchten Strategien. Die Verringerung der Bodenbearbeitungsintensität in den USA und der Anbau von Zwischenfrüchten, erwiesen sich als zweitgünstigste Strategie in Bezug auf die Kosten. Letzteres war in allen drei Fällen mit vergleichbaren Minderungskosten möglich. Die Unsicherheiten bei der Schätzung des Kohlenstoffbindungspotenzials führten jedoch zu großen Unterschieden zwischen den besten und den schlechtesten Szenarien, mit Ausnahme von Brasilien. Schließlich verursachte der Einsatz von Düngemitteln mit erhöhter Effizienz (Inhibitoren), der in den USA und in Brasilien möglich war, die höchsten Minderungskosten. Langfristig betrachtet stiegen die Minderungskosten der Maßnahmen mit Kohlenstoffbindung an und erreichten ein ähnliches oder höheres Niveau als die Maßnahmen ohne dieses Potenzial. Ausgenommen davon ist die Verringerung der Bodenbearbeitungsintensität, die in dieser Zeitperspektive geringere Kosten aufweist.

Zusätzlich wurde die Möglichkeit einer Kombination aller machbaren Minderungsstrategien in allen drei Regionen untersucht. Diese Kombination maximiert die erreichbaren Minderungspotenziale in jedem Fall. Das höchste Potenzial in der kurzfristigen Betrachtung konnte in den USA erreicht werden und war etwa viermal so hoch wie in den anderen Ländern. Dennoch waren die Kosten in den USA und in Brasilien vergleichbar, in Deutschland jedoch höher. Langfristig sind die Minderungspotenziale in den USA und in Brasilien ähnlich, aber die Kosten waren in Brasilien am niedrigsten, gefolgt von den USA und schließlich Deutschland.

Insgesamt wurden in dieser Arbeit die Auswirkungen der Einführung von Minderungsstrategien in Anbausystemen in einem realistischen Kontext analysiert, um umfassende Ergebnisse zu erzielen. Da in jedem Fall dieselbe Methodik angewandt wurde, sind die Ergebnisse vergleichbar und können in Zukunft durch Wiederholung des Ansatzes auf weitere Regionen und Kulturen ausgeweitet werden. Darüber hinaus sind die Ergebnisse zwar spezifisch für den Kontext, in dem sie berechnet wurden, sie können aber auch Erkenntnisse für vergleichbare Regionen liefern.

Table of contents

List of Figures

List of Tables

List of abbreviations

a	Annum
Avg.	Average
BMEL	German Federal Ministry of Food and Agriculture
BR	Brazil
C	Carbon
CAP	Common Agricultural Policy
CH_4	Methane
CO_2	Carbon dioxide
CO_{2eq}	Carbon dioxide equivalent
CONAB	Brazilian National Supply Company
DAP	Diammonium phosphate
DESTATIS	German Federal Statistical Office
DE	Germany
DüV	Fertilizer Directive
DWD	German Meteorological Service
EMATER	*Empresa Brasileira de Pesquisa Agropecuária*
EMBRAPA	Brazilian Agricultural Research Corporation
EPA	Environmental Protection Agency
EU	European Union
EUR	Euros
FAO	Food and Agriculture Organization
Gt	metric gigaton (1,000,000,000 metric tons)
GWP	Global warming potential
ha	hectares
IBGE	Brazilian Institute of Geography and Statistics
ICF	Inner City Fund (consulting company)
IFA	International Fertilizer Association
ILUC	Indirect land use change
IPCC	Intergovernmental Panel on Climate Change
JRC	Joint Research Centre

kg	kilograms
LFA	*Landesforschungsanstalt für Landwirtschaft und Fischerei Mecklenbug-Vorpommern*
LfL	*Bayerische Landesanstalt für Landwirtschaft*
MACC	Marginal abatement cost curve
MAGYP	Brazilian Ministry of Agriculture, Livestock and Fisheries
MAP	Monoammonium phosphate
MJ	Megajoule
N	Nitrogen
N_{min}	Mineral nitrogen available in the soil
N_2O	Nitrous oxide
NH_3	Ammonia
NO_3^-	Nitrate
NOAA	National Oceanic and Atmospheric Administration
NO_x	Oxides of nitrogen
NUE	Nitrogen-use efficiency
OECD	Organisation for Economic Co-operation and Development
t	metric ton
UAN	Urea ammonium nitrate
US	United States (of America)
USA	United States of America
USD	United States dollar
USDA	United States Department of Agriculture
WRI	World Resources Institute

1 Introduction

1.1 Background

The coming into force of the Paris Agreement in 2016 has emphasized the need to transition to a more sustainable society by reducing anthropogenic greenhouse gas (GHG) emissions to mitigate climate change. Agricultural production not only is severely threatened by climate change but also is relevant as it accounts for a significant share of global GHG emissions. In the case of crop production, the loss of soil organic carbon and, especially, the application of synthetic nitrogen fertilizers are, by far, the most significant sources of emissions (WRI, 2022). In this regard, the intensity of the utilization of fertilizers directly drives the release of GHG emissions. However, fertilizer rates can vary considerably among countries and regions, as they depend on agronomic, social and economic conditions. Hence, to assess the potential to reduce emissions from agricultural production, one needs to comprehend the technicalities and the feasibility of mitigation strategies at the farm level in order to identify the most economical alternative to abate GHG emissions.

Despite the known complexity of crop production systems and the uncertainties surrounding potential costs of adopting alternatives with lower emissions, research has focused on assessing primarily two aspects: the technical potential to lower GHG emissions with a specific mitigation strategy and the possible impacts of mitigation policies based on statistical models. Technical potentials for GHG reduction usually are estimated by measuring the emissions of an alternative practice and comparing them against those of the established one. These are usually estimated utilizing farm trials; e.g., Parkin and Hatfield (2014) and Abdalla et al. (2014). However, these trials are not necessarily representative of the agricultural conditions of the region and may not portray how farms are usually managed. Moreover, the costs, operational challenges and the overall implementation process in realistic farm conditions are rarely included in these studies. Consequently, determining realistic and representative mitigation costs is not feasible with the assumptions utilized, as key elements are not described or remain unaccounted for.

The impacts of possible mitigation policies are projected utilizing statistical models. These are designed to estimate the mitigation potential of a sector - e.g., agricultural production - affected by a mitigation policy. Averages of yields, input usage intensity, reduction potentials and costs, among other variables, are assumed to be valid for entire regions or countries and then fed into models of varying complexity. Examples of these estimations are the calculations by Schneider et al. (2007) and Tang and Ma (2022). However, these models consider the entire sector and rely mainly on fixed assumptions regarding the production system and farm operations. For instance, mitigation of emissions by adopting cover crops on a farm potentially can reduce fertilizer losses, affect the following crop and require additional machinery and inputs, among other aspects not typically considered in the assumptions of the model. Hence, these models simplify agricultural production and do not take into account the farm-level repercussions from interventions to crop production systems (Bakam et al., 2012).

Yet crop production systems are the result of a complex optimization of multiple variables. Local characteristics such as availability of machinery and labor, soil and climate conditions, crop rotation effects or changes in crop yield have subsequent impacts on farm productivity and profitability. Thus, these variables and their interactions define crop production in a specific region. Hence, they determine the feasibility and conditions in which mitigation strategies can be adopted. Moreover, these variables can be markedly different across regions. GHG mitigation strategies can be expected to differ accordingly, suggesting that considerable variation in mitigation potentials and costs may exist. Consequently, the regional agronomic, economic and technical characteristics of the farm must be taken into account in the evaluation of mitigation strategies.

However, while some of these variables may be available in statistical databases, this usually is not the case, especially sufficiently disaggregated to enable a detailed analysis. Moreover, assessing mitigation strategies also entails evaluating the repercussions that the strategies may have on farm operations and management. In this regard, regional growers and agricultural experts can provide estimates for the missing information (Ogle et al., 2013). Furthermore, such sources also have the necessary understanding of the regional context to evaluate the farm-level repercussions derived from the adoption of the strategies. Consequently, it is key to incorporate local expertise in order to generate a realistic analysis of the economics of GHG mitigation measures (Rosenzweig and Tubiello, 2007). To identify efficient and effective policies to reduce GHG emissions from crop production, a comprehensive evaluation of the farm-level feasibility and costs of mitigation strategies is necessary.

1.2 Research objectives

The main objectives of this thesis are to determine:

(1) What are the GHG mitigation potentials and costs derived from the implementation of farm-level GHG mitigation strategies in selected crops?

(2) How do these mitigation potentials and costs compare across strategies and farms in different regions?

1.3 Approach and structure of the thesis

The main approach used to answer the research questions is to utilize case studies. Considering limited time and funding, case studies enable the detailed analysis of mitigation strategies in crop production systems across different settings. These are realized in three different countries depicting diverse agronomic, climate and soil conditions as well as varying regulatory conditions. Each case study represents a relevant crop in a key agricultural region in a selected country. Based on this division, a general review of scientific literature is first conducted, and the review of region-specific information is integrated into each case study.

Chapter 2 provides the general background on GHG emissions from crop production and the common considerations and limitations on the research of mitigation strategies. The chapters begin by introducing the current global GHG emissions from all human activities. The geographical distribution of the emissions as well as the main sources from agricultural production are then shown. Then, the relevance of nitrogen utilization and soil management in crop production is reviewed as these are the main sources of emissions from this sector. Afterward, the common methodologies and approaches used to study GHG mitigation in agriculture are presented. The limitations of previous research are discussed to determine the main scientific contributions and novelties of this research project.

Chapter 3 presents the process used to define the case studies, followed by an explanation of the approach applied to assess the mitigation strategies and calculate their costs, emissions and mitigation potentials. First, the criteria specified for the selection of the crops and regions in which the case studies are conducted are explained. Afterward, the methodology utilized in each case study is introduced. First, the boundaries of the system and the functional unit are defined. For each case study, a Typical Farm is constructed based on a focus group discussion with growers and agricultural experts. The implementation of the focus group approach is explained stepwise. The results from the focus group are utilized to describe the key costs entailed in the production of the selected crop. Moreover, the focus group is used to evaluate the feasibility and implications of adopting the mitigation strategy in a realistic farm-level context.

Next, the methodology used for the calculation of the GHG emissions is presented. The methodology in this thesis includes an adjustment to the usual assumptions used to calculate nitrogen-driven emissions. The adjustment ensures that the nitrogen applied to the system matches the losses, which is necessary for the estimation of the mitigation strategies affecting nitrogen fertilization.

Finally, the preliminary list of GHG mitigation strategies evaluated for each Typical Farm is presented. The criteria used to determine the feasibility of implementing them is presented first. This includes an explanation on the approach used to depict the variation of the mitigation potentials and costs, which consists results in three different outcome levels: a standard case, best and worst cases. This is followed by a discussion of the possible technical, economic and agronomic implications of each mitigation strategy. The methodology used to calculate the mitigation potential of each strategy is presented. This discussion and methodologies are obtained from a literature review but are presented only for strategies that are evaluated in at least one of the case studies. Last, the approach to calculating the mitigation costs to provide policy advice is presented.

The three following chapters present the case studies: chapter 4 assesses corn production in Iowa, USA; chapter 5, corn in Paraná, Brazil; and Chapter 6, winter wheat in Mecklenburg-Vorpommern, Germany. The case studies are structured identically, as shown in chapter 3.9. Initially, the selection of the specific region within each country is presented. This is followed by a review of the context in which the Typical Farm operates and a description of the production system. This provides the baseline scenario against which the mitigation strategies are compared. Then, the strategies are discussed subsequently and independently from each other. The evaluation of the strategies finishes with the presentation of the resulting changes in costs, emissions and corresponding mitigation costs. This is followed by a comparison with the results found in scientific literature. The possibility to combine mitigation strategies is presented after the mitigation strategies have been discussed. The ranking of all the strategies assessed in the Typical Farm is presented, as well as a summary and main findings.

Chapter 7 compares the mitigation potentials and costs across the Typical Farms and discusses the main differences and similarities. A discussion regarding the methodology and overall application of the research concept of this thesis is shown in chapter 8 and the conclusions and advice for policymakers based on the findings of this thesis in chapter 9. The summary is provided in chapter 10.

2 Emissions from agriculture and research on mitigation strategies

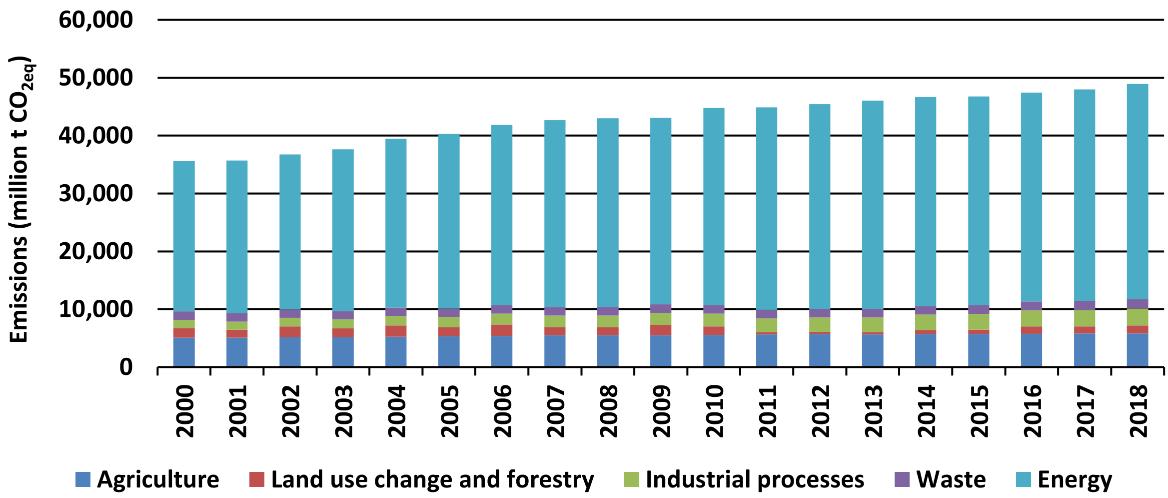
This chapter explains the GHG emissions from agricultural production and considerations on the possibilities to abate these emissions in crop production systems. This presents the general context in which this thesis is conducted. The chapter begins by presenting the geographical distribution and trends in GHG emissions from agriculture and its subsectors as well as from land use change. Nitrogen and soil management are the most relevant sources of emissions from crop production systems. The main factors driving the release from these sources at farm-level are introduced. Moreover, the main findings from key scientific literature and the common approaches used in the research on GHG emissions, mitigation potentials, and costs are discussed. Finally, the limitations identified in previous studies and overall research are stated.

2.1 Global GHG emissions and trends in agricultural production

Trends in global emissions and relevance of agriculture

Global anthropogenic GHG emissions have risen constantly for the past decades, as presented in Figure 2.1. The total output of emissions increased by 43% between 2000 and 2018, reaching 49 Gt CO_{2eq} (WRI, 2022). The emissions shown are divided based on the definitions of sectors provided by IPCC (2019a), which allocates the emissions to the sources where they are released. Most of the increase is in the energy sector, which depicts the emissions from electricity and heating production.

Figure 2.1 Global greenhouse gas emissions by sector from 2000 to 2018



Source: own elaboration based on WRI (2022).

The emissions from the agricultural sector, which includes the production of crops and livestock, increased by 14% between 2000 to 2018, totaling 5.8 Gt CO_{2eq}. In 2018, this sector represented approximately 12% of the total emissions, indicating it is the second largest source.

The sector "land use change and forestry" represents the emissions from deforestation, forest fires and changes in soil usage and management. This sector accounted for 3% of the global emissions in 2018. Changes in regulations in Brazil resulted in a significant a decrease in deforestation in the early 2010s, which lead to a substantial decrease in the emissions from this sector (Assunção et al., 2015). However, the expansion of agricultural land in Asia and Africa has sped up in the past decade (Jayathilake et al., 2021), meaning that, globally, emissions have remounted to the same level of the early 2000s.

The main driver of emissions from land use change and forestry worldwide is the expansion for agricultural use (Alexander et al., 2015). Therefore, it could be argued that the agricultural sector is directly responsible for an additional 3%, increasing the sector's share of emissions to 15% of the global total.

Adding the emissions from the manufacture of agricultural inputs, energy and transport, which are not accounted for in the sector "agriculture" in the emissions inventory shown in in Figure 2.1, increases the total contribution of agriculture to approximately 25% of total yearly anthropogenic emissions (Lamb et al., 2021).

It must be noted that these estimations are based on aggregated data used for emission reporting at the national or continental level. This contrasts with the farm-level approach utilized in this thesis to calculate the emissions from production systems.

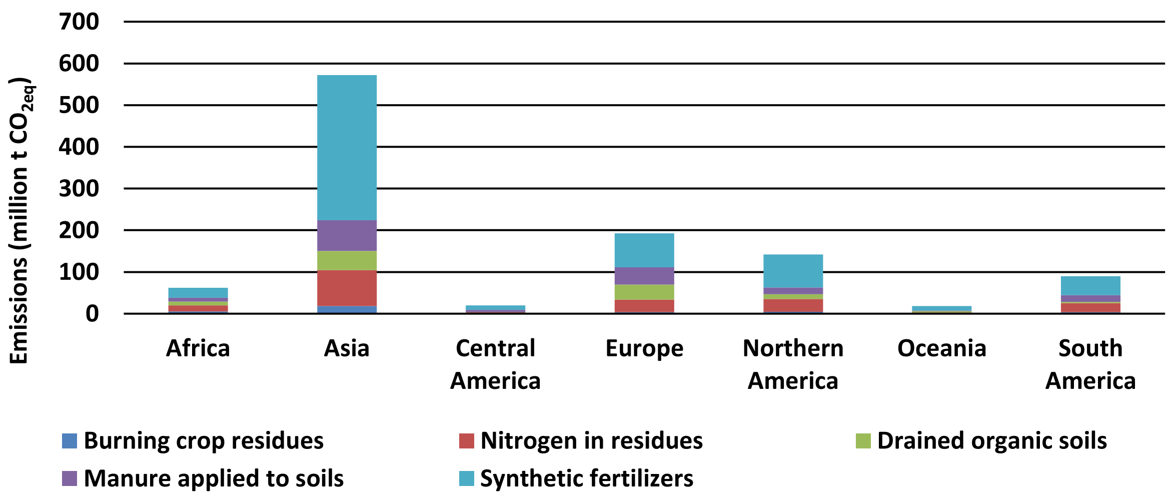
Emissions from agriculture and managed soils

The sector "agriculture and managed soils" can be divided into three categories based on the source of emissions: livestock, paddy rice and managed soils. Emissions from livestock production, resulting from enteric fermentation from ruminants and manure management, are mainly methane (CH_4) and, to a lesser degree, nitrous oxide (N_2O). These are approximately 68% of the sector's world total (FAO, 2022). Paddy rice production also releases methane, which equals 13% of the sector's total output. Emissions from managed soils are N_2O, released mainly from nitrogen applications. These account for 19% of the total. Over 90% of the emissions from managed soils can be attributed to crop production and the rest to grassland (IFA, 2001). Hence, crop production can be considered to be a relevant source of GHG emissions from agricultural production, as it represents approximately a fifth of the total from this sector.

According to FAO (2022), approximately 54% of the emissions from managed soils are released from synthetic nitrogen applications, 17% from nitrogen in crop residues and 15% from manure applied to soils. The N_2O emissions released from burning crop residues and draining peatland also are included in the managed soils category, but combined they represent less than 10%.

The shares of emissions from all categories in the agricultural sector have remained unchanged since 2000 based on FAO (2022). In 2018, Asia was the largest emitter of emissions from managed soils, with 53% of the total, followed by Europe with 18% and North America with 13% and South America with 8%. The emissions from all continents are presented in Figure 2.2.

Figure 2.2 Greenhouse gas emissions from managed soils by continent in 2018



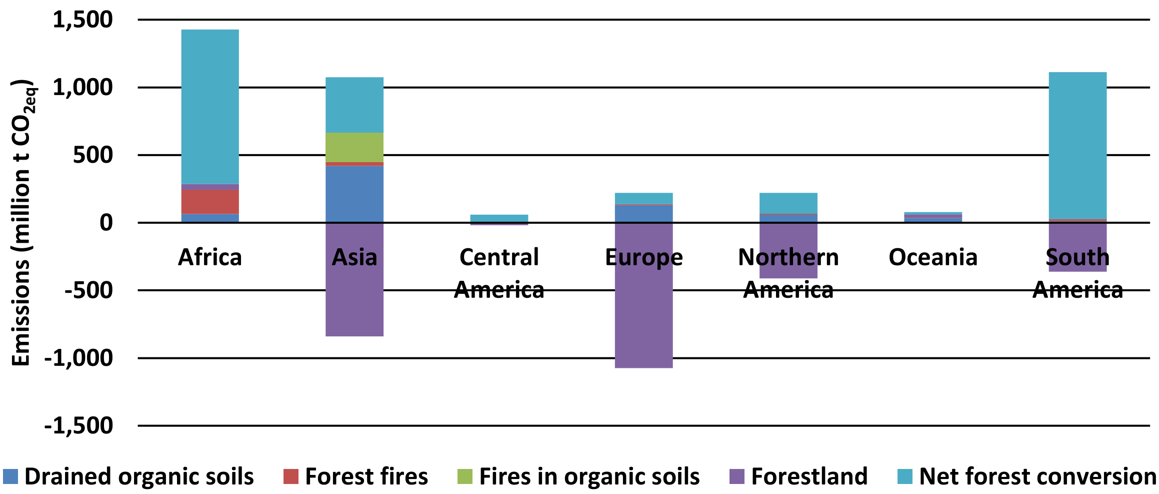
Source: own elaboration based on FAO (2022)

Emissions from land use change and forestry

The sector "land use change and forestry" emits carbon dioxide (CO_2). This sector has the particularity that it not only releases this gas but can also sequester it. The sequestration occurs mainly due to the incorporation of carbon in woody plants and soil organic matter, where it can remain bound for long periods (Deyn et al., 2008). This indicates that the values presented for this sector depict the emissions, the sequestration and the net change from emissions and sequestration.

In 2018, the emissions from this sector were approximately 4 Gt CO_{2eq} (Figure 2.3). These were driven by the net conversion of forestland to grassland or cropland, which accounts for 74% of that value (FAO, 2022). However, carbon sequestration occurs mainly in forestland and is estimated to be 2.6 Gt CO_{2eq}. Thus, the global net change in the sector land use change and forestry is a release of 1.4 Gt CO_{2eq}. Africa and South America are the largest net emitters, with 58% and 29% of the total, while Europe and North America are the only net sinks, with 82% and 18%, respectively.

Figure 2.3 Greenhouse gas emissions from land use change and forestry by continent in 2018



Source: own elaboration based on FAO (2022)

2.2 Nitrogen: emissions and relevance in crop production systems

Nitrogen can be considered among the most crucial nutrients for crop production (Eickhout et al., 2006; Kraiser et al., 2011; Maheswari et al., 2017). This input is crucial to attain high yields and increase farm profitability (Yadav et al., 1997). The discovery of the Haber-Bosch process to produce synthetic nitrogen was the main enabler of the increase in crop yields observed in the last century (Smil, 1999). This discovery more than quadrupled crop output and became the largest source of nitrogen to cropping systems (Stein and Klotz, 2016). It is estimated that the total nitrogen fertilizer applied in agricultural production increased 10 times since the 1950s (Robertson and Vitousek, 2009). Before the use of synthetic nitrogen, agriculture relied solely on biological nitrogen fixation from bacteria and the application of animal manure (Soumare et al., 2020). This restricted crop yields, indirectly limiting population growth because of the insufficient food supply (Sutton et al., 2011). Moreover, the dependence on fertilizers is expected to continue increasing as the global population is expected to reach 9 to 10 billion people in 2050 (Smith et al., 2013). Nonetheless, globally, there are considerable disparities in the crop yields attained, which can be attributed to the availability of nitrogen fertilizers (Drechsel et al., 2015). Currently, some regions still lack access to sufficient fertilizer to ensure consistent yields and food supply (Mosier, 2004).

The increased availability of nitrogen fertilizer has led to overuse, promoting losses, and severe disruptions of natural ecosystems from pollution (Melillo, 2021). The recovery rate of the nitrogen applied in cereals - i.e., nitrogen use efficiency, is estimated to be only 33% on a global average (Omara et al., 2019). This coefficient varies greatly across regions and has not increased over the past 15 years, despite several international initiatives (Lassaletta et al., 2014). The low recovery rate implies that a considerable share is lost to the environment as gas and via leaching to underground water bodies (Bowles et al., 2018). These gaseous losses occur as N_2O, ammonia (NH_3), oxides of nitrogen (NO_X) and nitrate (NO_3^-) if lost as leaching. Additionally, these nitrogen compounds can affect the ecosystem by causing the eutrophication of water bodies, promoting the loss of biodiversity and generating soil acidification (Bouwman et al., 2002).

Emissions from nitrogen in agricultural systems

In the context of GHG emissions, the most important nitrogen-based gas is N_2O (Reay et al., 2012). Agriculture accounts for 76% of global man-made N_2O emissions, with 58% of these emissions coming from managed soils mainly for crop production (WRI, 2022; IFA, 2001). This gas is released by the activity of soil microorganisms, which are essential for the maintenance of the ecosystem. Bacteria take up this nutrient and utilize it in their metabolism, releasing N_2O as a by-product (van Spanning et al., 2005). This occurs naturally and is promoted by, among other factors, the availability of nitrogen, especially from synthetic sources (Venterea et al., 2005; Bouwman, 1996).

Crop residues also are a substrate for the bacteria responsible for these emissions. The magnitude of the emissions from residues depends on the species of plant, its development state and amount of residues (Aulakh et al., 1991). The overall release of N_2O by bacteria is determined by the source of the nitrogen, soil temperature and moisture and oxygen saturation (Stott et al., 1986; Stein and Klotz, 2016). The considerable number of factors affecting the rate of release of the gas also implies that estimating emissions is a complex process with a significant variability in the values calculated. Thus, the definition of mitigation strategies derived from these estimations also shows this large variability.

Since these emissions are part of the nitrogen cycle, it can be concluded that N_2O from crop production is at least partially unavoidable as emissions are a part of the system (Yan et al., 2014). Still, these losses are over-proportionately increased if an oversupply of nitrogen occurs, which can be interpreted as a fertilizer rate that is higher than the combined requirements of the crop and the soil bacteria as a function of yield (Grant et al., 2006). Moreover, the production of fertilizers, especially nitrogen-based, is an energy-intensive process (Brentrup et al., 2018), releasing additional emissions that can be allocated to crop production. The emissions from the manufacture can be more than one-fourth of the emissions released by the use of the fertilizer. Hence, adjusting the nitrogen rate to match the requirements of the crop and thereby minimize emissions is indispensable to abate the effect of agriculture on climate change. Furthermore, the oversupply of nitrogen represents an inefficient input use, implying unnecessary costs for the farm, decreasing its profitability (Flaten et al., 2019).

2.3 Carbon: emissions and relevance in crop production systems

Soil can be regarded as a finite natural resource that provides several ecological functions necessary for crop production (Kopittke et al., 2019). It is the medium where plants develop, it stores and regulates water flow, and it acts as a buffer and storage for nutrients (Komatsuzaki and Ohta, 2007). The soil's composition and properties are determinants of nitrogen availability and the plant’s growth potential (Li et al., 2014). Thus, the preservation of its physical, chemical and biological qualities is fundamental to avoid limiting plant development and yields. Still, it is estimated that 22% of global agricultural acreage is degraded (Jie et al., 2002).

Soil degradation implies a decrease in soil organic carbon, which is released to the atmosphere as CO_2 (Follett, 2001). The carbon share of the soil organic matter is referred to as soil organic carbon and is estimated to be approximately 58% (Stockmann et al., 2013). In turn, soil organic matter portrays the sum of the microbial and plant biomass, humus and organic particulate material. Soil organic matter helps maintain yields and is crucial in the cycling of nutrients and water, critical elements in the preservation of ecosystems. Carbon in the soil is constantly being cycled: It is released as emissions via oxidation and replenished by inputs of plant biomass (Chapin et al., 2009). Alternations to the losses and inputs in this cycle can lead to net permanent changes in the soil carbon content. The loss of soil organic carbon also implies the release of N_2O emissions, which contrarily to carbon emissions, cannot be recaptured by an increase in the soil carbon stock (Liu et al., 2019).

Long-term land use management as well as the physical properties of the soil largely define soil condition at the farm level (Pulleman et al., 2000). In this regard, the two main determinants of soil degradation and carbon content in cropping systems are tillage and crop residue management. Tillage breaks and mixes soil layers to prepare the seedbed for the sowing of the crop, facilitates root growth and controls weed populations, among other factors critical to attaining high yields (Ohiri and Ezumah, 1990; Meade and Mullins, 2005). However, intensive tillage operations such as soil inversion have exacerbated soil degradation by exposing more layers to erosion (Hussain et al., 2021). Crop residues can be incorporated into the soil to promote decomposition and return nutrients to the soil. Hence, the removal of crop residue and biomass for use as feed or generation of biofuels promotes soil degradation, as the replenishment of the soil's naturally lost carbon is lowered (Blanco-Canqui and Lal, 2009). Nonetheless, the influence of tillage and crop residue management in soil carbon is the subject of considerable debate. While previous findings indicated that these practices generated substantial changes in the carbon stocks, newer studies have reported markedly lower effects.

Reversing the practices that led to carbon loss in agricultural soils has the potential to increase soil organic carbon (Lal, 2004a). The addition of practices aimed at increasing the output of biomass, such as cover crops or cash crops with high straw generation, can help potentiate this effect (Reicosky, 2018). The input of carbon in crop residues is linearly correlated with soil organic carbon (Huggins et al., 1998). This indicates carbon sequestration, which can be interpreted as the capture of atmospheric carbon. The carbon is absorbed into the plant's structure via photosynthesis and can be later incorporated into the soil's carbon pool (Johnson et al., 2007; Komatsuzaki and Ohta, 2007). This increases soil organic matter, increasing yields, water-holding capacity and nitrogen availability (Quiroga et al., 2006). Nonetheless, compared with losses, carbon accumulation is slow and the effects that climate and soil properties have on the accumulation rate still are not yet understood (Grandy and Robertson, 2007). The potential of the soil to capture carbon is finite, as the soil no longer binds additional carbon when it has reached a new equilibrium (Powlson et al., 2011). The captured carbon can be lost if the practice that led to its sequestration is no longer utilized by the farm. Thus, a long-term strategy at the farm level is required to attain the possible economic and environmental benefits from increased soil organic carbon (Lal, 2010). Consequently, for this thesis, the assessment of the strategies leading to changes in carbon content is conducted taking into account the long-term adoption to ensure the attainment of the GHG reduction potential.

2.4 Research on GHG mitigation strategies in agricultural systems

Estimation of mitigation potentials and the adoption of strategies

The calculation of mitigation potentials and costs can be divided based on the limitations assumed to adopt the strategies. The technical mitigation potential represents the maximum biophysical reduction achievable assuming simultaneous implementation of all the technically feasible strategies, regardless of the limitations or implementation costs to adopt them (Moran et al., 2013). Globally, for the combined agricultural and land use change sectors, this potential is estimated to be 5.5-6 Gt CO_{2eq} per year (Smith et al., 2008), which would decrease the total anthropogenic emissions from 2018 by more than 10% (WRI, 2022). However, social and political limitations may restrict the realization of mitigation strategies since they may not be considered acceptable in the given context (Wreford et al., 2017). Moreover, the strategies may increase the costs of the production system without increasing revenues, reducing the acceptance that producers have.

The economic mitigation potential depicts the annual emission reduction in relation to a defined carbon price considering adoption costs but without accounting for socio-cultural or institutional barriers (IPCC, 2014). This potential usually is expected to be lower than the technical potential and is normally calculated using different carbon prices as reference. For instance, the economic potential of global carbon sequestration has been calculated to be 1.5 Gt CO_{2eq} at a carbon price of 20 USD/t CO_{2eq} and 2.6 Gt CO_{2eq} at a carbon price of 100 USD/t CO_{2eq} (Smith, 2016). The wide range in the calculations is the result of the variation in the assumptions of the respective authors, especially regarding the likely opportunity costs entailed in the adoption of the strategies. Moreover, it can be expected that there is a significant discrepancy between the actual adoption costs at the farm level and the assumptions used in these estimations.

Several of the strategies assessed by the authors of these studies include substantial changes to the agricultural systems, integrating or switching livestock and crop production. Thus, it is not possible to allocate mitigation potentials to specific sources. Still, most strategies in crop production focus on the mitigation and sequestration of CO_2 via soil management and reduction of N_2O emissions from fertilizer applications. These gases and sources usually are the focus of the mitigation strategies because of their total contribution to the sector's emissions (Snyder et al., 2009).

Nonetheless, there are several limitations to realizing the mitigation strategies in the agricultural sector (Lankoski et al., 2020). At the farm-level, the actual or perceived economic burden from the adoption of the strategy can be considered among the most significant limitations (Wreford et al., 2017). This burden can result from increased costs, additional investment, or a reduced yield, which would also increase food insecurity. Other possible barriers are a lack of information and access to it, transaction costs and social and cultural aspects (Smith et al., 2007; Wreford et al., 2017). Furthermore, the technical and economic preconditions necessary for the implementation may limit the adoption of mitigation strategies, especially in developing countries, as the access to inputs and financing may be limited (Tubiello et al., 2009). Farm-specific attributes such as size and structure as well as age, education and farmer's experience also are determinants of the willingness to adopt strategies (Dessart et al., 2019). Consequently, it can be inferred that calculation of mitigation strategies must take into account the farm and the grower's conditions to generate realistic results.

The regulatory and institutional frameworks of the country can be barriers as well. Transparent and enforceable land ownership regulations are essential to incentivize the adoption of strategies (Palmer, 2011). Although not directly attributable to the farm, an insufficient institutional capacity to monitor and support the mitigation strategies can slow their adoption. The monitoring also implies an accurate estimation of the emissions and mitigation potentials, which entails that region-specific formulas and coefficients must be generated (Mbow et al., 2012).

The mitigation of emissions in the agricultural sector should be achieved without negatively influencing the yields. Assume a GHG mitigation strategy that abates emissions but also reduces crop yields by a greater amount; e.g., emissions are lowered by 5% but yield decreases by 10%. This indicates that each ton of production proportionately releases more emissions. Thus, fulfilling the demand for the product would actually release more emissions than before the adoption of the mitigation strategy. Additionally, a reduction in crop yields may generate a gap in the supply of the product. Thus, elsewhere, land must be transformed by growers to produce the missing yield or the productivity in existing land must be increased.

This is likely to promote deforestation, releasing CO_2 and possibly offsetting a significant share of the abatement attained (Pan et al., 2020). Moreover, it is possible that the missing yield produced elsewhere may be supplied by comparatively less efficient growers. For instance, they could apply more fertilizer to produce a ton of product compared with the growers who adopted the initial mitigation strategy. The geographical displacement of emissions as a result of a mitigation strategy is known as "emission leakage" in the context of carbon accounting or as "Indirect Land Use Change" (or ILUC) in the bioenergy sector[1] (Ostwald and Henders, 2014). A similar effect can be expected if crops are substituted to mitigate emissions - for instance, legumes are grown instead of cereals. In this case, the magnitude of the emissions from ILUC depends on the substitutability of the products (Gnansounou et al., 2008). Overall, quantifying the impact of ILUC is a complex task (Broch et al., 2013). Nonetheless, estimations from previous research can provide insights applicable to this thesis to quantify the effect that ILUC could have.

Win-win scenarios

In the context of GHG mitigation, win-win scenarios depict strategies that not only lower emissions but also generate additional benefits to the producers or surrounding ecosystems (Smith et al., 2007). Possible benefits could be an improved yield, increasing the grower's revenue and increasing food security (Follett et al., 2005; Lal, 2004a). Practices promoting carbon sequestration usually increase soil fertility and productivity as well as the resilience of the system against extreme weather events (Moraes Sá et al., 2014; Lal, 2014). Increasing nitrogen efficiency in cropping systems reduces NO_3 leaching into water bodies, decreasing eutrophication (Conley et al., 2009).

Win-win scenarios that reduce emissions while increasing profitability have negative mitigation costs. Usually, strategies entailing a reduction in tillage intensity or optimization of nitrogen fertilization result in this type of scenario. In reality, however, farmers are reluctant to implement the win-win scenarios calculated despite the apparent economic improvement (OECD, 2010). An explanation for this counterintuitive option could be that the profitability calculations are not adequately defined. For instance, they may omit relevant costs or overestimate revenues. Additionally, non-economic factors also can influence the grower's decision to adopt a mitigation strategy. Thus, win-win mitigation strategies must be assessed thoroughly as they may be the result of limitations of the model used in the calculations. This topic and the implications for these case studies are discussed in chapter 8.

Nonetheless, the implementation of the mitigation strategies can have negative impacts, such as a decrease in biodiversity due to higher usage of crop care products (Bustamante et al., 2014; Gardner et al., 2012) or an increased release of mercury and sulfur into the atmosphere from additional fossil fuel requirements (Elbakidze and McCarl, 2007). Pure win-win scenarios are rare as most GHG mitigation strategies require a trade-off (DeFries et al., 2004).

Approaches to calculating and comparing mitigation costs

In a context of limited resources and time, identifying the most efficient mitigation strategies is essential to developing climate policy. This requires understanding the abatement potential and the costs implied in the realization of the strategies (MacLeod et al., 2015; Fellmann et al., 2021). The implementation of a mitigation strategy could entail costs such as the monitoring and enforcing of the strategy (transaction costs) as well costs that may arise from an increase in the market prices of the goods, among others. The calculation of these economic costs generally is conducted using statistical modeling. At the farm level, the cost of the strategy is the sum of economic changes the grower perceives compared with a

---

[1] For the remainder of this study, the terms emission leakage and Indirect Land Use Change are used as interchangeable synonyms.

business-as-usual scenario. For instance, these can result from increased expenditure or reduced income; i.e., foregone revenue. In this regard, this thesis calculates the farm-level costs derived from the adoption of the strategy.

The mitigation potential represents the net change in emissions caused by a respective strategy initiated. The mitigation cost - e.g., cost in USD to reduce one ton of CO_2 - can be interpreted as the monetary cost incurred to attain a reduction in GHG emissions (Povellato et al., 2007). In scientific literature, mitigation costs also are referred to as mitigation cost-efficiency or cost-effectiveness.

The mitigation costs are used as a metric to compare different mitigation strategies. Individual mitigation strategies can be compared graphically using the Marginal Abatement Cost Curve (MACC). A MACC depicts the strategies in a two-dimensional plot in which each column represents a specific mitigation approach (Fellmann et al., 2021). The width of the column signifies the annual abatement potential in tons of CO_2 and the height the mitigation cost per ton of CO_2. Normally, the strategies (or columns) are arranged in order of abatement cost. Theoretically, the sum of the widths of all the columns indicates the total GHG mitigation potential achievable in the system assessed if all the strategies are implemented simultaneously.

Normally, mitigation costs and potentials in MACCs are calculated using one of two approaches: top-down or bottom-up. The top-down approach is based on calculating sectoral or microeconomic models to compute the impact of policy interventions, such as setting a carbon price, taking into account the economy as a whole (van Vuuren et al., 2009; Fellmann et al., 2021). Usually, these econometric models rely on macroeconomic aggregated data as well as a predefined set of conditions for mitigation strategies. These models consider changes in the market derived from the strategies selected - e.g., prices of inputs and output. The models calculate which combination of strategies would be adopted by the farms based on their interactions and competition (Vermont and Cara, 2010). This implies that the result is a scenario in which the strategies cannot be interpreted independently of each other but rather as an element within the scenario. The mitigation potentials and costs are estimated for an entire region or economic sector and the results are not dividable. Examples of MACCs can be found in the works by Pérez Domínguez et al. (2020) and Schneider et al. (2007).

The bottom-up approach focuses on the technical implementation, costs and reduction potential of each individual strategy evaluated first at farm level (Fellmann et al., 2021). This approach usually does not account for changes in market conditions or its driving forces. The farm-level results can be scaled up to estimate the effect on a region or country. The bottom-up approach has the advantage that it enables the interpretation of the strategies individually, but the results cannot be aggregated because interactions between strategies are not accounted for. Interactions between strategies may significantly affect the economics and mitigation potential (MacLeod et al., 2015; Fellmann et al., 2021). Hence, this approach is not implicitly adequate to assess the combined mitigation potential of several individual strategies, unless the interactions are evaluated. As examples, Pellerin et al. (2017) and ICF (2013) used a bottom-up approach to construct MACCs.

2.5 Contribution of this thesis to literature

Limitations of previous research

Significant limitations have been identified in the calculation of mitigation alternatives, either as individual strategies or in the context of MACCs. Examples of the limitations are the nonrepresentation of adoption barriers, unclear system boundaries and the omission of complex synergies or interactions between the individual strategies; e.g., competition for machinery (Kesicki and Ekins, 2012; Kesicki and Strachan, 2011). Key limitations generally omitted are the availability of inputs, machinery and labor in the farms. Furthermore, studies typically evaluate crops in an isolated context, meaning that the effect of the crop rotation, the preceding and following crops, are not normally taken into consideration, especially in farm-level studies. These elements may impose substantial constraints on the adoption of mitigation strategies because of the competition for machinery, management of crop residues and soil, or shared plant diseases, among others.

Additionally, most studies rely solely on highly aggregated, publicly available databases; thus, they are limited by the availability of statistical data or the level of disaggregation (Ragnauth et al., 2015). Usually, activity data is not covered in these databases; e.g., specific fertilizer applications for a given crop. Additionally, the difficulty in harmonizing different databases can also hinder their usability for mitigation calculations (Eory et al., 2018). Hence, distinctions for specific crops are omitted or are based on assumptions rather than on actual information. Consequently, the mitigation strategies are assumed to be fixed and identical for entire regions or countries, ignoring the effect of the variability in climate, soil and farm structure.

Hence, it could be argued that previous studies using only statistical data have been dependent on an oversimplified depiction of the productive system[2] and the local context. Thus, they are lacking a comprehensive understanding of the agronomic and economic logic of existing crop production systems and their implications for mitigation strategies.

Arguably, conducting case studies partially tackles these limitations. This methodology facilitates a detailed holistic assessment of a complex situation and enables the combination of the expertise of stakeholders plus statistical information a (Morland et al., 1992). Case studies can be used to reveal and explain complex causal links (Yin, 1994). Hence, case studies can be used to assess the farm-level limitations and implications that the adoption of a mitigation strategy may have in order to generate realistic results.

In the context of case studies, focus groups can be conducted with growers and agricultural experts to describe a representative crop production system. This has the advantage that it incorporates their expertise in describing the farm. Focus groups with farmers are a frequent approach to evaluate mitigation strategies and understand the limitations to their adoption (Stuart and Schewe, 2016; McCarl et al., 2021; Kahiluoto et al., 2012; Swinton et al., 2015). The focus group approach enables the assessment of the grower's reaction to a change in the framework conditions; for instance, new policies or market conditions (Feuz and Skold, 1992; Chibanda et al., 2020). The participants have the opportunity to elaborate on complex ideas, facilitating a more comprehensive description (Reid et al., 2007). In addition, the integration of diverse stakeholders with divergent stances within a focus groups fosters discussion by eliciting the exchange of ideas (Kowarsch et al., 2016). Furthermore, the grower's management decisions to implement a mitigation strategy and the drivers of these decisions can be perceived as qualitative data. This type of information usually is not included in statistical datasets unless specific surveys have been conducted. It has been recognized that the participation of stakeholders in studies concerning climate change mitigation in agriculture is essential to create meaningful policy advice (Dooley et al., 2018).

Moreover, in previous farm-level studies evaluating more than one strategy or bottom-up MACCs, the analysis of synergies in a combination of strategies rarely was included. Thus, the theoretical maximum mitigation potential and resulting cost from a joint implementation of strategies is not revealed.

A comparison of the mitigation potential and costs from different studies is not adequate in every case, as the methodologies, system boundaries and factors considered can differ (Eory et al., 2018; Kesicki and Ekins, 2012). The restricted comparability of the results from diverse studies limits their usefulness in

[2] To illustrate, while several databases with fertilizer rates applied are available, they rarely present the data on the specific fertilizer used or the time of application. Thus, a detailed analysis - for instance to assess the competition for machinery - is not feasible.

policymaking (Vermont and Cara, 2010). Taking into account that most studies focus on one region or country, the direct comparison of mitigation costs across farms on different continents is limited.

Main contributions of this thesis to literature

Considering the limitations of previous research on GHG mitigation strategies and costs in crop production, this thesis tackles the following issues:

(1) Derivation of comprehensive and representative farm-level results: By actively involving growers and agricultural consultants in a focus group discussion, this thesis integrates the intricacies and challenges of crop production in the selected regions utilizing case studies. This approach provides a realistic baseline describing the production system. The approach also is suitable for generating and evaluating feasible mitigation scenarios, which account for the technical, agronomic and economic limitations.

(2) Assessment of the combination of strategies: The interactions and possible limitations to adoption derived from the joint implementation of multiple strategies are evaluated. Based on the evaluation, a realistic maximum mitigation potential and corresponding costs are calculated.

(3) Comparability of mitigation potentials and costs across different regions: The utilization of a standardized methodology to calculate costs and emissions and to assess the feasibility to implement the strategies enables a direct comparison of the results in different contexts.

2.6 Relevance of farm-level mitigation costs for policymakers

Design of mitigation policies in agricultural context

In addition to farm-level mitigation costs, as calculated in this thesis, policymakers must consider further factors to design efficient environmental policies. Mitigation strategies may generate tradeoffs or synergies that can be positive or negative (Verspecht et al., 2012). For instance, lowering the nitrogen rate to abate emissions may also reduce the eutrophication of water bodies (Huang et al., 2017), but could threaten food security if yields are affected (Zhang et al., 2013). Thus, interactions of the policies must be evaluated. Transaction costs also must be taken into account to evaluate the efficiency of different environmental policies (McCann, 2013). These costs depend on several factors, such as the specific measure, existing infrastructure and information, legal systems and possibility to combine policies, among others (Coggan et al., 2010). Thus, the share of transaction costs in the total cost of a policy can vary considerably. To illustrate, Coggan et al. (2010) reviewed literature studying transaction costs in environmental policies and found that these range from 21% to 50% of the total cost of the policy. Rørstad et al. (2007) estimates the share of transaction costs in agricultural environmental policies in Norway to be approximately 20%. Arguably, this reveals that farm-level costs can be expected to represent the largest share of the total cost of the mitigation policy. Moreover, transaction costs tend to become lower with technological development and the adoption of digital infrastructures (Ehlers et al., 2021), meaning that the share of farm-level costs may become higher. Consequently, while farm-level costs alone are not sufficient to derive comprehensive mitigation policies, understanding these costs compliments the development of efficient abatement policies by providing essential data (Sánchez et al., 2016).

The farm-level mitigation costs calculated in each case study are ranked in ascending order. This list provides policymakers with an initial indication of which strategies should be prioritized when further evaluating the GHG mitigation policies. It must be noted that this list of priorities is valid only for the share of the farm population used as reference for the assessment in each case study; i.e., the Typical Farm. Thus, the ranking also should be used as an indication for which strategies should be further researched, not only to verify the findings from this thesis and increase their accuracy, but also to be able to expand the analysis to additional farms, production systems and regions.

Time horizon of the mitigation policies

The economics of the mitigation strategies in agriculture may vary based on the time horizon considered for the policy. In this regard, policymakers must differentiate between strategies reducing emissions and strategies promoting carbon sequestration (Fellmann et al., 2021). Reducing GHG emissions by adopting a new technology or changing a practice - e.g., lowering the nitrogen rate to reduce nitrogen-driven emissions - will imply a reduction of GHG emission compared to the baseline scenario as long as the practice is in use. Therefore, assuming all costs and processes remain unchanged, the cost-efficiency of this mitigation strategy is constant.

In comparison, carbon sequestration is a finite and reversible process. This means that the practice that led to binding the carbon in the soil - e.g., growing cover crops to incorporate the biomass into the soil - must be used continuously despite no longer generating a net sequestration. Hence, strategies with carbon sequestration presents two stages: an initial finite transition stage with carbon sequestration and a following stable stage in which the practice is perpetually in use, but no net additional carbon is sequestered (see chapter 3.5.6). Assuming the yearly costs of the strategy are constant, it can be inferred that the cost-efficiency of policies promoting carbon sequestration worsens when the time horizon considered is extended, because a higher proportion of years without carbon sequestration must be accounted for.

Therefore, in order to provide advice for policymakers, the mitigation strategies must be evaluated considering different time horizons. Hence, the ranking of strategies in this thesis is divided using a short-term and a long-term perspective (see chapter 3.8.). It must be noted that, given the goal of the Paris Agreement to reach global net zero emissions by 2050 and the expectation that agriculture substantially contributes towards this target (Leahy et al., 2020), it is advised that policymakers prioritize the mitigation strategies using the short-term perspective.

3 Methodology and research concept

This chapter introduces the methodology used to assess the mitigation strategies in this thesis. First, the criteria used to select the crops, countries, and specific regions are explained, including the delimitation of the system considered in the analysis. The case study in each region is based on a Typical Farm described using focus groups. Both concepts and their application to evaluating the implementation of the GHG mitigation strategies in this thesis are presented.

The costs considered in the Typical Farm and the methodology used to calculate the emissions are introduced. Last, the criteria applied to select the mitigation strategies discussed in the focus groups, the description of the strategies selected, and the approach used to calculate the mitigation potential are explained.

3.1 Selection of crops and regions for case studies

Existing literature and studies were reviewed to obtain an indication of possible regions and crops for the case study. The principle was to gather information on the mitigation potential and costs achievable in crop production systems in an international comparison. However, no applicable database presenting this standardized information disaggregated at sufficient levels of detail could be identified. Thus, the selection of the crop and regions for the case studies is based on criteria deemed crucial to generate impactful results.

The first criterion is that the crop and region must be relevant for global food security and agricultural markets. Although the scalability of the results is not evaluated, the representativity of the conditions in which the strategies are derived is evaluated in detail. Thus, selecting a relevant crop and region would imply that the results are valid on a larger scale. The total production of the crop is used as a proxy for this aspect.

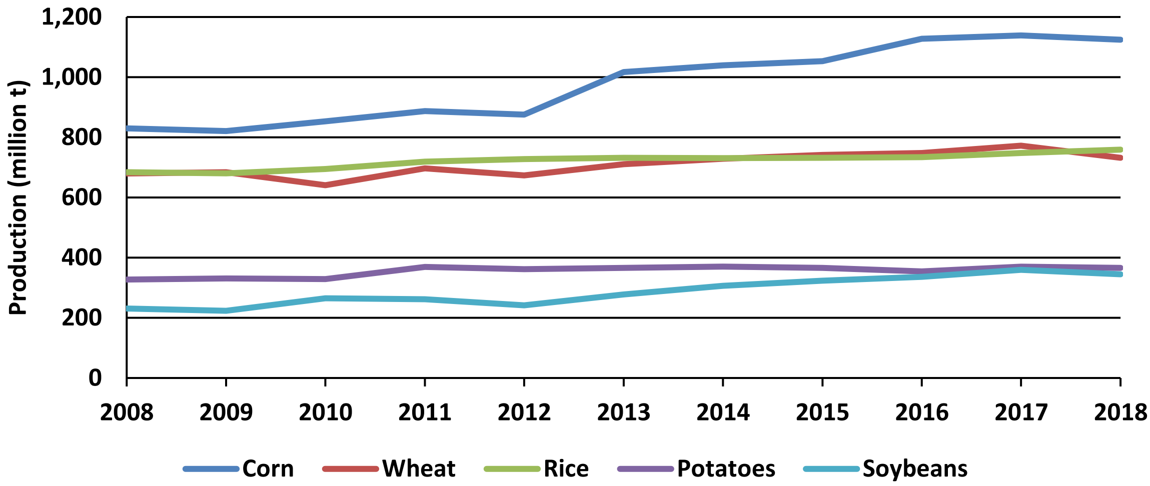
The second criterion is that the network *agri benchmark* Cash Crop (see chapter 3.4) must have scientific partners in the country of the case study. This is fundamental for the assessment of the mitigation strategies since the partners not only provide region-specific know-how but also can support the organization of the focus group used to discuss the strategies with stakeholders. This is considered fundamental to deriving realistic results adequate to the local context.

The application of these two criteria can be divided into two stages. The first is the choice of crops and countries where the case studies are conducted, which is explained in this chapter. The second stage is the selection of the specific region within the country where the Typical Farm of the case study is located. This assessment is conducted in the corresponding chapter in each case study.

Selection of crops and countries for the case studies

The five most-produced arable crops from 2008 to 2018 are used as the first criteria in the selection and are presented in Figure 3.1. Globally, corn is the most-produced crop. Its production shows an upward trend and has increased by approximately 35% since 2008, which is a significantly larger increase compared with the other crops presented, except for soybeans, which are less relevant in absolute terms. Wheat and paddy rice production are similar in volume for the entire period covered. The global production of these crops was approximately 35% lower than that of corn in 2018. However, paddy rice production differs considerably from other crops - in particular, due to the intricacies and particularities of its water management system. Thus, it is deemed to be beyond the scope of this thesis. Potatoes and soybeans are the fourth and fifth most-grown crops, but compared with wheat and corn, they can be considered less relevant. Hence, corn and wheat are deemed as potential crops for the case studies.

Figure 3.1 Most-produced crops globally from 2008 to 2018

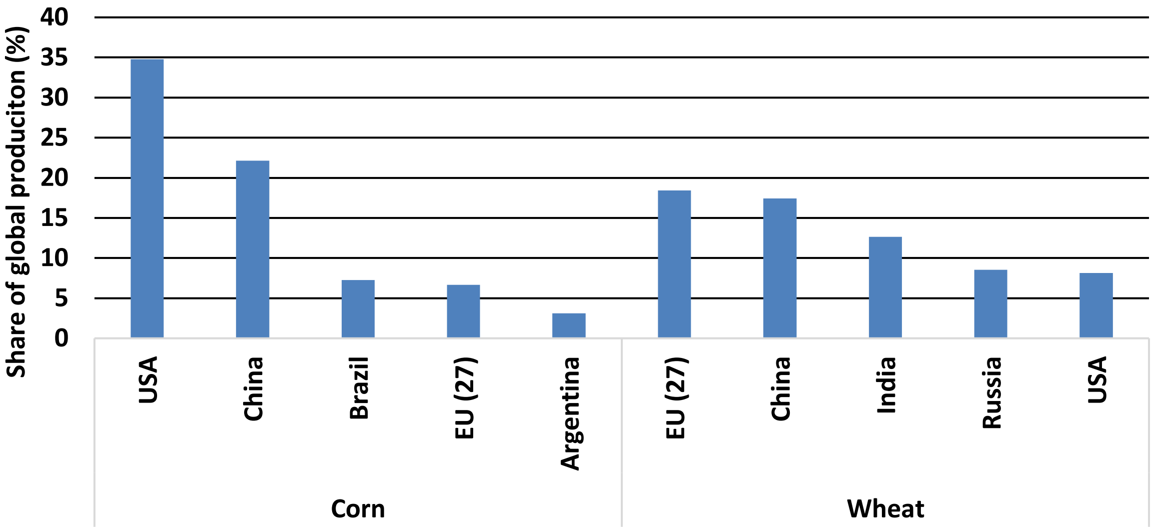


Source: own elaboration based on FAO (2022)

The largest producers of corn and wheat are presented in Figure 3.2. The USA is the largest producer of corn, with approximately one-third of global production, followed by China (22%) and Brazil (7%). Inquiries were sent to selected members of the *agri benchmark* network in these countries, in which the project and research concept were presented to determine the possibility of obtaining their support. Further aspects such as time availability, language barrier and acquaintance with the Typical Farm and focus group approaches also were considered based on the feedback provided by the researchers. Based on these considerations, cooperation with scientific partners in the USA and Brazil could be established.

The European Union (EU) is the largest wheat producer, with 18% of global total output. Based on the average from 2008 to 2018, France is the largest grower, with 29% of the EU output, followed by Germany with 19%. Conducting a case study in France likely would imply a language barrier as the researcher doing this thesis does not speak French. Thus, it is not guaranteed that the interviews and discussions can be conducted without the help of a qualified translator, which would increase the complexity of the organization of the case studies and increase the risk of missing key information due to errors in the translation. Additionally, it must be taken into account that this research project is funded and carried out in a German institution. Moreover, the German Federal Government launched the Climate Action Programme 2030, which aims at reducing the emissions from the country by 55% by 2030. The agricultural sector is expected to contribute toward this goal by lowering its emissions by 14% compared with 2014. Consequently, the last case study in this project is conducted for wheat in Germany. This both helps compare mitigation costs at an international level for a relevant producer and provides insights that can contribute to the efficient adoption of the German government's Climate Action Programme 2030.

Figure 3.2 Largest producers of corn and wheat by share of total production – average 2008-2018



Source: own elaboration based on FAO (2022)

3.2 System boundaries

The agricultural activities and inputs necessary to produce the selected crop in the field are defined as the system assessed for the mitigation strategies. Based on this delimitation, the costs considered are the crop establishment costs (seeds, fertilizers, and crop protection), liming, and the costs entailed in the realization of the farm operations. Within this category, diesel usage, labor required, and the cost of the machinery (depreciation and maintenance) are taken into account. The use of agricultural contractors to conduct field operations is considered as well. These elements are assumed to depict the crucial field-level costs that can be affected by the implementation of the mitigation strategies.

GHG emissions released from the field resulting from the use of these inputs as well as the emissions driven by the crop are considered in this thesis. Thus, the emissions included are the soil emissions from nitrogen additions (fertilizers and crop residues), the CO_2 from urea applications and liming, and diesel for the field operations. The carbon emissions from changes in the soil organic carbon resulting from changes in the farm operations are accounted for in the assessment. Additionally, the carbon footprints entailed in the manufacturing of the fertilizers and the diesel are addressed as well, as, without crop production, these emissions would not occur[3]. As with costs, the emissions from these elements are deemed as those most likely to be affected by the mitigation strategy.

Emissions not considered in the system

The emissions from the manufacture of selected elements used in crop production, other than fertilizer, are not considered in this thesis. The total carbon footprint from the manufacture of machinery has been estimated to represent only approximately 6% of the total emissions in intensive crop production (Pugesgaard et al., 2015). Similar results have been found for the manufacture of herbicides and pesticides (Chen et al., 2018; Audsley et al., 2009; Pugesgaard et al., 2015) as well as for seeds (Camargo et al., 2013). Moreover, only the change in the usage of these inputs resulting from the mitigation strategy is needed to determine the mitigation costs (see chapter 3.7). In other words, only the GHG emissions entailed in the

[3] Neither the emissions from diesel manufacture and usage nor from the manufacture of fertilizers are included in the GHG inventories from the sector agriculture following IPCC definitions, as presented in chapter 2.1.

farm operations or inputs that differ from the baseline scenario are relevant. Therefore, it is inferred that the difference in GHG emissions from changes to these inputs would be comparatively small and their effect on the overall mitigation potential is considered negligible.

3.3 Functional unit

For this analysis, the costs are expressed in dollars (USD), and emissions in carbon dioxide equivalents (CO_{2eq}). These are expressed per hectare, which is used as the functional unit. Costs and emissions are used to calculate the cost to mitigate one ton of GHG emissions.

Additionally, in the case that the mitigation strategy generates a change in the crop yield, the emissions are also calculated per ton of product. This reveals whether there is a net increase in global GHG emissions derived from releasing more emissions per unit produced (see chapter 2.4).

The gases considered in this analysis are carbon dioxide (CO_2) and nitrous oxide (N_2O). Although methane (CH_4) also is described in the methodology from IPCC (2019b), none of the inputs or processes included in this thesis release CH_4.

CO_2 and N_2O have different intensities as GHGs; thus, they are standardized using the Global Warming Potential (GWP). This is the most common metric used to measure the climate impact of different gases (Timma et al., 2020; Holtsmark, 2015). It is a calculated coefficient in which the warming capacity of a specific gas depends on its potential to absorb energy and its lifetime in the atmosphere. The cumulative annual effect of the gas is compared against CO_2, which is used as a reference in a ratio. This ratio is averaged over a period of time, typically 100 years. The resulting value for the gas is expressed as CO_{2eq}. Hence, this can be interpreted as the amount of CO_2 needed to equate the climate-effect of a specific gas over 100 years. The values published in the Fifth Assessment report by IPCC (2013) are used and indicate a GWP for N_2O of 296 kg CO_{2eq}.

3.4 Definition of Typical Farm and costs

The concept of Typical Farms as defined by *agri benchmark* are used for the evaluation of the GHG mitigation strategies. The *agri benchmark* Cash Crop network is composed of agricultural economists and experts, covering a variety of agricultural systems across the globe. The network is coordinated at the Thünen Institute of Farm Economics in Germany.

A Typical Farm describes the dominant farm type and production systems in a relevant agricultural region, which is indicated to produce the bulk of the agricultural output (Zimmer and Deblitz, 2005). Detailed information on crop rotations, mechanization, farm operations, crop yields, input use and other agronomic factors are depicted for each Typical Farm (Hemme et al., 1997). These data are gathered following a standardized procedure, including data collection and validation. This ensures consistency and comparability across Typical Farms. The detailed data described and procedures utilized are presented in Zimmer and Deblitz (2005).

However, in reality, the Typical Farm does not exist. It is rather a construction based on data gathered in a focus group discussion in which agricultural consultants and growers from the selected region participate. Consequently, the Typical Farm is deemed to be representative, albeit not in the statistical sense, as it does not describe the average of the statistical data (Zimmer and Deblitz, 2005). Nonetheless, in cases when this type of data is available, it is used for a comparison with the information provided by the participants of the discussion.

The results from focus groups are processed with the TYPICROP model. The model calculates relevant agronomic and economic indicators for each crop and for the whole farm. The formulas and fundamental assumptions entailed in this model are described in detail in the thesis conducted by Nehring (2011). Chibanda et al. (2020) present a comprehensive analysis and explanation of the Typical Farm methodology. It must be noted that the standardized procedure used for Typical Farms in the *agri benchmark* network has been adapted for this thesis. The detailed approach utilized in the case studies is presented in the next chapter.

3.4.1 Focus group discussion

For this thesis, focus groups with growers and agricultural experts are used to describe the Typical Farm in the baseline situation as well as the changes required to adopt the mitigation strategies. The approach is presented in this chapter and is applied in each region where a Typical Farm is established. The regions are selected using the criteria previously presented.

3.4.1.1 Preparation of focus group discussion

The guidelines for the focus group discussion in the context of *agri benchmark* were established by Nehring (2011) and were utilized in the works of Krug (2013), Walther (2014) and Dehler (2023)

Preparation of focus group

First, cooperation with a scientific partner in the region selected must be established. This facilitates the realization of the focus group approach for the Typical Farm. Members of the *agri benchmark* network in the selected countries and regions are contacted. The project, objectives and expectations regarding their support are explained in detail. If they are not available to participate, a recommendation for possible candidates to take this role is requested. This process repeats until the support of a local scientific partner has been gained.

The researcher conducting this thesis and the local scientific partner jointly search for participants for the focus group. An invitation outlining the project and the objective of the discussion is prepared. The invitation is sent to selected farm managers chosen by the scientific partner, who are asked for voluntary participation. Additionally, an agricultural consultant is required for the session, who receives the same invitation. However, depending on their stipulations, the consultant receives payment for participating. The participants are not required to have any specific knowledge of GHG mitigation strategies or to bring any type of farm-specific information other than their expertise. Thus, the basic focus group is composed of the researcher conducting this thesis, the scientific partner, three to five farm managers and the agricultural consultant. The exact number of participants is presented in the corresponding chapter in each case study.

Draft of the Typical Farm

Parallel to the search for participants, a draft of the Typical Farm for the case study is constructed. In the cases where *agri benchmark* already has a Typical Farm in the region where the case study takes place, this Typical Farm is reviewed for the draft. This draft is complemented using statistical data and scientific literature. If no Typical Farm is available in the region, only statistical data and literature are used to define the draft. The draft includes a description of the overall Typical Farm used in the cost and emission calculation of the case studies:

(1) Acreage (ha)

(2) Crop rotation (including cover crops)

(3) Tillage management

(4) Crop yields for all crops produced (t/ha)

(5) Moisture content of the harvested parts (%)

(6) Labor and input intensity and costs for the crop selected for the assessment (see chapter 3.4.2)

This data portrays what is assumed to be a normal year, based on the averages from the past three seasons. This draft ensures that all focus group participants start from the same initial baseline, which can then be adjusted. The main assumptions and questions of the draft are discussed with the local scientific partner before the session. A handout and presentation detailing the agenda for the discussion as well as the mitigation strategies are jointly prepared to structure the discussion (Kühn, 2018).

A literature review on the possible mitigation strategies in the selected region is conducted to prepare for the focus group discussion (see chapter 3.6). Preliminary calculations regarding possible changes at the farm level are conducted using scientific literature and the draft of the Typical Farm. The values calculated are used in the discussion to provide indications of the magnitude of the changes that are expected.

3.4.1.2 Beginning of session and definition of baseline scenario

The researcher and the scientific partner moderate and take notes, which are also recorded if the participants agree. The recordings are used to complete the notes and are deleted afterward. The session begins with an introductory round during which participants introduce themselves and provide background on their experience and farm. The objective of the session is then presented to the participants. The concept of Typical Farm as defined by *agri benchmark* is introduced and, if possible, the preliminary results from the previous case studies are presented and a discussion takes place. Afterwards, the draft of the Typical Farm is debated. All the elements of the draft are discussed until a consensus on the assumptions is reached among the participants, including the reference period used to define a normal year. The result of this discussion depicts the Typical Farm assumed as the baseline scenario of the case study. This ensures that all the participants start from the same initial situation. The representativity of the results is evaluated in the corresponding chapter of the case study.

3.4.1.3 Implementation of mitigation strategies

The mitigation strategies are introduced by the researcher and are debated in successive order. A brief explanation of how the strategy mitigates emissions and examples of the possible implementation are provided. The participants are invited to ask questions about the strategy. Afterward, the discussion begins, starting with how the farm managers assume the strategy would be implemented on the Typical Farm. Each adjustment proposed is discussed, which may entail changes to the inputs and practices already conducted or the inclusion of new ones. Detailed data on machinery, diesel and labor requirements are gathered for changes in farm operations. The discussion continues until a consensus on all changes has been reached. The participant’s estimation of potential repercussions in yields, soil and other agronomic factors are included in the assessment. Previous research has shown that subjective and potential controversial assessments of the participants usually lead to a consensus after a short time (Nehring, 2011). The discussion then proceeds to the next mitigation strategy, and the process repeats until all the strategies have been addressed. A final round of questions is opened for all participants, the summary of the discussion is presented, and, lastly, the participants are thanked for their contributions.

After the focus group discussion is completed, the researcher and scientific partner meet to compare notes and clarify the key messages and insights. These results are validated with scientific literature and additional selected interviews are conducted to extend the analysis. The results of these processes are presented in the assessment of each mitigation strategy.

3.4.2 Costs considered

The list of inputs and costs in the Typical Farm discussed in the focus group for the baseline scenario and the mitigation strategies entails the following elements:

(1) **Seed:** cost per hectare

(2) **Fertilizer:** type, rate and unitary cost (per product)

(3) **Pesticides**: sum of herbicide, insecticide, and fungicide application costs per hectare

(4) **Liming**: type, rate and unit cost

(5) **Diesel:** liters required for the operations and cost

(6) **Labor**: hours required for the operations and cost

(7) **Machinery**: depreciation and repairs as explained in chapter 3.4.3

(8) **Contractor:** total costs per hectare

These costs are used for the basic description in the case studies, but additional categories not previously included can be added or existing elements modified, based on the input from the focus group.

3.4.3 Machinery costs

The calculation of machinery costs in the Typical Farm approach is explained in detail in the work by Nehring (2011), which is adopted in this thesis. Only the machinery costs derived from changes to the farm operations necessary for the mitigation strategies are calculated. To illustrate, this calculation would be done in case equipment is no longer required or it is changed; e.g., replacing a plough with a cultivator. The calculations are conducted utilizing the formulas presented in this chapter based on the data from the focus group discussions.

Cost calculation

The machinery costs account for the depreciation of the machine. The depreciation costs are divided into two subcategories: depreciation based on historical prices (Equation 1) and depreciation based on repurchase prices (Equation 2). Considering depreciation based on repurchase prices corrects for possible distortions between farms with different inflation rates (Nehring, 2011).

$$\boldsymbol{Dep_{hist}} = \frac{\boldsymbol{Purchase - Salvage}}{\boldsymbol{Depreciation\ period}} \qquad (1)$$

$$\boldsymbol{Dep_{rep}} = \frac{\boldsymbol{Repurchase - Salvage}}{\boldsymbol{Depreciation\ period}} \qquad (2)$$

Dep_{hist}: depreciation of machine based on historical prices (USD/a)
Dep_{rep}: depreciation of machine based on repurchase prices (USD/a)

Purchase: historic purchase price of machine (USD)
Salvage: salvage value after depreciation (USD)
Repurchase: repurchase price for the same machine after depreciation (USD)
Depreciation period: depreciation time of the machine in years

Additionally, the financing costs of each machine are included in the depreciation costs. These are obtained from the average fixed capital in the machine (Equation 3). Finance costs have two subcategories: finance costs for debts in the machine (Equation 4), which represents the cash debt; and the opportunity costs from the equity in the machine (Equation 5), which represents the interest rate that could have been earned if a deposit would have been made.

$$\boldsymbol{FixedCap} = \frac{(\boldsymbol{Purchase} - \boldsymbol{Salvage})}{\mathbf{2}} + \boldsymbol{Salvage} \qquad (3)$$

$$\boldsymbol{DebtCap} = \boldsymbol{FixedCap} \times (\mathbf{100\%} - \boldsymbol{Equity}) \times \boldsymbol{i_l} \qquad (4)$$

$$\boldsymbol{OppCap} = \boldsymbol{FixedCap} \times \boldsymbol{Equity} \times \boldsymbol{i_d} \qquad (5)$$

FixedCap: average fixed capital in machine (USD/a)
DebtCap: cash cost debt on the machine (USD/a)
Equity: equity share in fixed assets (%)
i_l: interest rate long term loan (%)
OppCap: opportunity cost in equity of the machine (USD/a)
i_d: interest rate long term deposit (%)

The total depreciation cost of each machine in the Typical Farm is calculated with Equation 6.

$$\boldsymbol{Dep_{mac}} = \boldsymbol{Dep_{hist}} + \boldsymbol{Dep_{rep}} + \boldsymbol{DebtCap} + \boldsymbol{OppCap} \qquad (6)$$

Dep_{mac}: total annual depreciation cost of the machine (USD/a)

The annual expenditure on repairs and maintenance also is accounted for in the Typical Farm. These are added to the depreciation costs to depict the annual machinery costs. This value must be distributed to allocate the costs per hectare, which is analog to the cost per pass. A distinction between types of machines is necessary for this allocation.

For towed equipment, the distribution of machinery is based on the acreage on which it is utilized every year (Equation 7). However, a tractor's depreciation is usually distributed based on the yearly running hours, as the same tractor can be utilized for different operations. Thus, the capacity of the tractor to conduct a specific operation be taken into account when allocating costs (Equation 8).

$$\boldsymbol{MachineCost_{towed}} = \frac{\boldsymbol{Dep_{mac}} + \boldsymbol{Repairs}}{\boldsymbol{Util_{towed}}} \qquad (7)$$

$$\boldsymbol{MachineCost_{tractor}} = \frac{\left(\frac{\boldsymbol{Dep_{mac}} + \boldsymbol{Repairs}}{\boldsymbol{Util_{tractor}}}\right)}{\boldsymbol{Capacity}} \qquad (8)$$

$MachineCost_{towed}$: machinery costs for towed equipment (USD/ha)
$Util_{towed}$ annual utilization of the towed equipment (ha/a)

$MachineCost_{tractor}$:	machinery costs for tractors (USD/ha)
$Util_{tractor}$	annual utilization of the tractor (h/a)
Capacity	acreage that can be covered per hour during the operation (ha/h)

The combination of tractor and towed equipment necessary for the operations is obtained from the focus group. The data necessary for the calculations are obtained from the focus group and the preexisting Typical Farms and are shown in the Appendix (Table A.11 for the case study in the USA (Iowa) and Table A.12 for Germany (Mecklenburg-Vorpommern)).

Assessment of depreciation costs

The calculation of depreciation costs depends on the context in which the change in the farm operations occurs. The type of equipment used, competition for machinery and the total annual utilization, among other factors, are taken into consideration in the evaluation. Thus, depending on these aspects, the depreciation costs arising from the changes in operations may be assumed to be zero. This indicates that only a redistribution of the costs takes place, but no net change occurs at the farm level. This assessment is conducted each time a change in the operation occurs, which is explained in the chapter evaluating the mitigation strategy. In contrast, repair costs are assumed to be applicable for each pass.

Machinery costs in the baseline scenario

The total machinery costs in the baseline scenario are not necessary for the calculation of the mitigation costs, as only the changes derived from the implementation of the strategies are required. Nonetheless, a comparison between the Typical Farm constructed for the case studies and the preexisting Typical Farm from the *agri benchmark* network is conducted if data are available. If deemed similar, the machinery costs of the selected crop in the preexisting Typical Farm are implemented in the case study in the baseline scenario. These provide complementary information to understand the local context.

3.5 Calculation of GHG emissions

This chapter explains the methodology used to calculate the GHG emissions from the Typical Farm in the baseline and mitigation scenarios. Key considerations for the assessment of the mitigation strategies also are presented. The coefficients and their sources used in the formulas presented in this chapter are shown in the Appendix in Table A.2.

The climate regime is necessary for the calculation of emissions. This information is obtained from scientific literature presented in the baseline scenario of each case study.

3.5.1 Emissions from nitrogen

The methodology for the calculation of the emissions driven by nitrogen, which is usually used in this type of assessment, is provided by IPCC (2019b). Yet, the coefficients included in the methodology, in many cases, are aggregated and may not necessarily apply to the conditions of the Typical Farms. Therefore, it is assumed that the methodology must be modified, which is explained in chapter 3.5.1.2.

3.5.1.1 Direct emissions

According to IPCC (2019a), the category "direct emissions" (Equation 9) represents the N_2O released from the field where the crop is produced. Microorganisms take up the nitrogen from the soil and utilize it in their metabolism, releasing N_2O as a result (van Spanning et al., 2005). Emission intensity is governed by the availability of nitrogen (Venterea et al., 2005; Bouwman, 1996). Consequently, the more nitrogen added to the soil, the higher the release of N_2O will be if no other factors are changed.

The result of Equation 9 indicates the nitrogen emitted as N_2O. To transform this value into kilograms of N_2O, the result must be multiplied by the ratio of the weight of the nitrogen atoms contained in the N_2O molecule. This ratio is assumed to be 44/28 (IPCC, 2019a).

$$\boldsymbol{N_2O\text{-}N} = (\boldsymbol{F_{SN}} \times \boldsymbol{EF_{SN}}) + (\boldsymbol{F_{CR}} \times \boldsymbol{EF_{CR}}) \quad (9)$$

N_2O-N:	amount of nitrogen released as nitrous oxide (kg N_2O-N/ha)
F_{SN}:	input of synthetic nitrogen fertilizer (kg N/ha)
EF_{SN}:	emission factor for nitrous oxide from synthetic nitrogen input (kg N_2O-N/kg N/ha)
F_{CR}:	input of nitrogen from crop residues (kg N/ha)
EF_{CR}:	emission factor for nitrous oxide from nitrogen from crop residues (kg N_2O-N/kg N)

In general terms, an emission factor depicts the share of the input that is released as GHG. The previous version of the guidelines from IPCC (2006) provided one default emission factor for direct emissions. However, the refinement from IPCC (2019a) includes disaggregated coefficients based on the source of the nitrogen and the climate regime.

Calculation of nitrogen in crop residues

Following IPCC (2019a), the nitrogen in crop residues (F_{CR}) can be determined as a function of the crop's dry matter (Equation 10).

$$\boldsymbol{F_{CR}} = \boldsymbol{AG_{DM}} \times \boldsymbol{NC_{AG}} + \boldsymbol{BG_{DM}} \times \boldsymbol{NC_{BG}} \quad (10)$$

AG_{DM}:	dry matter of crop residues above-ground (kg DM/ha)
NC_{AG}:	nitrogen content in above-ground residues (kg N/kg DM)
BG_{DM}:	dry matter of crop residues below-ground (kg DM/ha)
NC_{BG}:	nitrogen content in below-ground residues (kg N/kg DM)

Based on IPCC (2019a), the dry matter harvested is calculated with Equation 11, which is required to determine the dry matter of above-ground residues (Equation 12).

$$\boldsymbol{DM_{harvest}} = \boldsymbol{Yield} \times (\boldsymbol{100\%} - \boldsymbol{Moisture}) \quad (11)$$

$$\boldsymbol{AG_{DM}} = \boldsymbol{DM_{harvest}} \times \boldsymbol{slope} + \boldsymbol{intercept} \quad (12)$$

$DM_{harvest}$:	dry matter of harvest (kg DM/ha)
Yield:	yield of the crop (kg/ha)
Moisture:	moisture content of the grain (%)

Slope and intercept are crop-specific coefficients obtained from IPCC (2019a). The residues below-ground can be calculated with Equation 13.

$$BG_{DM} = (DM_{harvest} + AG_{DM}) \times RS \qquad (13)$$

RS: ratio of below-ground to above-ground biomass

3.5.1.2 Adjustment factor

As previously mentioned, it is deemed necessary to utilize an adjustment factor for the calculation of N_2O emissions. This factor affects the indirect emissions, which are partly introduced in this chapter. The formulas for indirect emissions adopted in this thesis are shown in chapter 3.5.1.3.

The adjustment factor is not a part of the methodology from IPCC (2019a). Thus, the arguments for the need to adopt the factor and its assumptions are introduced and the calculation of the factor is presented afterward.

Nitrogen inflows and outflows

The nitrogen in the soil could be interpreted as a closed system that is permanently losing and receiving the nutrient. The nitrogen outflows are the nitrogen losses that occur based on IPCC (2019a) guidelines. These losses are via the previously presented direct N_2O emissions and through two indirect pathways.

The first of these pathways is the volatilization of nitrogen as NH_3 and as NO_X. These gases also are released as part of the nitrogen cycle. The second pathway is when nitrogen is transported away from the field via leaching and runoff[4]. If the precipitation transports the nitrogen as overland water, it is considered runoff; if the water infiltrates the soil pores and transports the nutrient below the reach of the roots it is labeled as leaching. This washed-off nitrogen is relocated to water bodies. Based on IPCC (2019a), these losses are calculated with Equation 14 and Equation 15, respectively.

$$N_{VOL} = F_{SN} \times Frac_{GASF} \qquad (14)$$

$$N_L = (F_{SN} + F_{CR}) \times Frac_{LEACH} \qquad (15)$$

N_{VOL}: amount of nitrogen lost from the soil via volatilization (kg N/ha)
$Frac_{GASF}$: fraction of synthetic nitrogen that volatilizes as NH_3 and NO_X (kg N volatilized/kg N)
N_L: amount of nitrogen lost from the soil via leaching (kg N/ha)
$Frac_{LEACH}$: fraction of nitrogen input that is leached (kg N leached/kg N)

IPCC (2019a) provides a disaggregated coefficient to be used for the fraction of nitrogen volatilized ($Frac_{GASF}$). However, this disaggregation is based on the type of fertilizer and does not account for climate. Another coefficient is provided for the nitrogen leached ($Frac_{LEACH}$). A single coefficient is provided that covers all the regions where leaching occurs[5].

However, nitrogen also is exported from the system in the harvested crop parts (Equation 16), which is not accounted for in IPCC's methodology. As with crop residues, the harvested portion of the crop also contains

[4] For the remainder of this study, the term leaching refers to the combination of nitrogen losses from leaching and runoff.

[5] According to IPCC (2019a), leaching occurs in regions where the climate is moist and wet, which have the same coefficient for nitrogen leached. In regions with dry climate, leaching is assumed to be zero.

nitrogen in its structure, which depends on the crop and region (Baligar et al., 2001). This value is retrieved from scientific literature. Thus, this can be interpreted as an additional nitrogen outflow from the system. Hence, the three outflows in the Typical Farm are direct N_2O losses, nitrogen losses from indirect pathways (leaching and volatilization) and the harvested crop.

$$\boldsymbol{N_{harvest} = DM_{harvest} \times NC_{HP}} \qquad (16)$$

$N_{harvest}$: amount of nitrogen exported via harvest (kg N/ha)
NC_{HP}: nitrogen content in harvested parts (kg N/kg DM)

The main inflow of nitrogen to the system is the application of nitrogen as fertilizer. The biological nitrogen fixation from legumes, either from the previous cash crop or from a cover crop, can be considered as another inflow. A share of the nitrogen fixated by legumes can become available to the following crop via mineralization (Ciampitti and Salvagiotti, 2018).

A further potential inflow of nitrogen is the nitrogen deposited from volatilization, which, in part, is available to the crop (Frink et al., 1999). To provide a reference, Zhang et al. (2015a) indicate that the average hectare of cropland in the USA receives 11 kg N/ha from deposition. It could be argued that on a regional level, a field receives the same amount of nitrogen as deposition that it loses as volatilization. However, other processes such as the combustion of fossil fuels also release nitrogen-based gases that are later deposited (Mosier et al., 2013). This implies that the nitrogen deposited is very different regionally as it depends on complex soil and climate processes and surrounding human activities, which, combined, limit its determination (Frink et al., 1999; Smil, 1999). Furthermore, the nitrogen deposited also is subject to the nitrogen cycle, indicating that not all of it is available. Considering the intricacies associated with nitrogen deposition calculation and that the grower has no influence on this variable, the nitrogen inflow from deposition is assumed as a constant and is not accounted for in the assessment. Therefore, the nitrogen inflows in the Typical Farm are from fertilizers and nitrogen fixation.

Initial theoretical nitrogen balance in the Typical Farm

To facilitate the explanation for the need for the adjustment factor, selected coefficients and results calculated in the case study in the USA are utilized in the following explanation. The explanation to obtain these values is provided in the baseline scenario of the Typical Farm in Iowa (Chapter 4.5.1) and the Appendix in Table A.2.

The crop rotation in the Iowa Typical Farm is soybean followed by corn, which is the crop used in the analysis. The nitrogen rate for corn equates to 181 kg N/ha. Soybean fixates nitrogen, but region-specific data indicate that this legume does not generate a nitrogen credit from biological fixation. Hence, the inflow of nitrogen from this source is assumed to be zero and fertilizers are the sole inflow.

The outflow of nitrogen as direct emissions is approximately 4 kg N/ha. Using the default coefficients from IPCC (2019a) without the adjustment factor reveals that 15 kg N/ha are lost via volatilization and 68 kg N/ha via leaching. Thus, the outflow of nitrogen via indirect losses totals 83 kg N/ha. The outflow via harvested parts is estimated to be 141 kg N/ha. Consequently, the total nitrogen outflow of corn in this Typical Farm is 227 kg N/ha. Theoretically, this indicates that the production of corn on this Typical Farm results in a net loss of 46 kg N/ha; i.e., the nutrient is significantly undersupplied. A prolonged undersupply of nitrogen would lower the soil fertility, reducing the crop's yield (Brentrup and Palliere, 2010).

Yet, as previously discussed, a Typical Farm is said to be a farm that produces the bulk of the agricultural output. To retain productivity, the grower must plan an adequate nitrogen supply to maintain yields and profitability in the long-term. It can be inferred that the grower has the know-how to produce the crop ensuring these aspects and he would have noticed if the nitrogen supply was not sufficient. Consequently, it could be argued that it is not realistic that the Typical Farm is undersupplying nitrogen, as theoretically indicated by this calculation. This result can be attributed to some of the general coefficients and assumptions used in the emission estimation.

Arguably, some emission coefficients portray the context of the Typical Farms. In general terms, the nitrogen content of fertilizers is constant and intrinsic to the type of fertilizer; thus, it is not affected by external factors. As in the case of Iowa, data on nitrogen fixation from legumes is generally available for several relevant agricultural regions. Likewise, the nitrogen exported via harvest is calculated using region and crop-specific values. Similarly, the direct losses are calculated using an emission factor that is disaggregated by source of nitrogen and climate. Consequently, it can be deemed that the inflow of nitrogen via fertilizer and the outflow via harvest and direct losses can be adequately calculated for the conditions of the Typical Farms.

On the contrary, the nitrogen outflow from leaching and volatilization are estimated using the coefficient from IPCC (2019a). This does not account for differences in climatic conditions. Both are biochemical processes that are affected by factors such as temperature (Li et al., 2020) and precipitation (Gu and Riley, 2010; Klocke et al., 1999).

Moreover, the coefficient for leaching losses ($FRAC_{LEACH}$) is estimated by IPCC based on measurements in grassland and arable land, which are averaged. The losses in these systems can be considerably different (Fraters et al., 2015). Similarly, leaching rates from organic and synthetic nitrogen are combined, which also can have markedly different loss rates (Di and Cameron, 2002). The large variation in the values utilized in IPCC's determination of the leaching coefficient is reflected in the uncertainty range. In this case, along with the default 24% adopted in this estimation, a range from 1% to 73% with a 95% confidence interval is provided.

Considering these aspects, it can be inferred that the leaching and volatilization coefficients do not properly depict the losses occurring in the Typical Farm. This is especially relevant as these losses amount to a significant share of the total nitrogen outflows calculated. Arguably, these explain the undersupply of 46 kg N/ha between the nitrogen inflow and outflow.

Estimation of adjustment factor

To adapt the leaching and volatilization coefficients, an adjustment factor is implemented to balance the nitrogen budget in the Typical Farm. It is assumed that nitrogen inflows equal the outflows, meaning that the nitrogen applied to the cropping system is lost by the end of the season. In other words, on a seasonal basis, the nitrogen stock in the soil remains unchanged with no net gain or loss. In each case study, this assumption is reviewed in scientific literature and modified if necessary.

As previously explained, the nitrogen inflows, as well as the direct emissions and the export via harvest, are deemed to be adequately estimated for the Typical Farm. Thus, these are used to calculate the nitrogen remaining in the system that is lost via leaching and volatilization (Equation 17).

$$N_{remaining} = N_{supply} - N_{harvest} - N_2O\text{-}N \qquad (17)$$

$N_{remaining}$: nitrogen in the soil susceptible to being leached or volatilized (kg N/ha)
N_{supply}: sum of nitrogen inflows (fertilizer and biological fixation) available to the crop (kg N/ha)
$N_{harvest}$: amount of nitrogen exported from the system via harvest (kg N/ha)
$N_2O\text{-}N$: amount of nitrogen released as nitrous oxide (kg N_2O-N/ha)

The adjustment factor can then be calculated with Equation 18.

$$\boldsymbol{AF = \frac{N_{remaining}}{N_{VOL} + N_L}} \quad (18)$$

AF:	adjustment factor for losses as leaching and volatilization in the Typical Farm
N_{VOL}:	amount of nitrogen lost from the soil via volatilization (kg N/ha)
N_l:	amount of nitrogen lost from the soil via leaching (kg N/ha)

The adjustment factor is used as an additional element that modifies the calculation of the N_2O emissions from leaching and volatilization. With this adjustment, the sum of the nitrogen losses from leaching and volatilization pathways will equate to the nitrogen in the system that is susceptible to being lost. Hence, the nitrogen balance in the Typical Farm is zero. Its integration into the methodology from IPCC (2019a) is explained in the next chapter, which introduces the formulas as implemented in this thesis.

3.5.1.3 Indirect emissions

Indirect emissions from atmospheric deposition

Almost all of the gases volatilized as NH_3 and NO_x from the field (N_{VOL}) are returned as deposition on other fields and water bodies (Schlesinger and Hartley, 1992), where they are susceptible to being emitted as N_2O. IPCC (2019a) guidelines and the adjustment factor are used to estimate these emissions (Equation 19).

$$\boldsymbol{N_2O_{(ATD)}\text{-}N = N_{VOL} \times AF \times EF_{ATD}} \quad (19)$$

$N_2O_{(ATD)}$-N: (kg N_2O-N/ha)	amount of nitrogen lost as nitrous oxide from atmospheric deposition
AF	adjustment factor for losses as leaching and volatilization in the Typical Farm
N_{VOL}:	amount of nitrogen lost from the soil via volatilization (kg N/ha)
EF_{ATD}:	emission factor for nitrous oxide from nitrogen deposited (kg N_2O-N/kg N)

As with direct emissions, the result of the equation is multiplied with a ratio of 44/28 (IPCC, 2019a) to transform the nitrogen into N_2O.

Indirect emissions from leaching and runoff

A share of the nitrogen lost via leaching (N_L) is released as N_2O. These losses are estimated based on IPCC (2019a) and the adjustment factor.

$$\boldsymbol{N_2O_{(L)}\text{-}N = N_L \times AF \times EF_L} \quad (20)$$

$N_2O_{(L)}$-N:	amount of nitrogen released as nitrous oxide from leaching (kg N_2O-N/ha)
N_L:	amount of nitrogen lost from the soil via leaching (kg N/ha)
EF_L:	emission factor for nitrous oxide from nitrogen leached (kg N_2O-N/kg N)

The factor 44/28 (IPCC, 2019a) is used to transform the result into N_2O.

3.5.2 Lime application and extraction

Agricultural lime is applied to the soil to reduce soil acidity and facilitate the absorption of certain nutrients by the plant. This amendment usually is not applied every year but, rather, once in the rotation, as the effect on soil chemistry can last several seasons. Lime contains carbon-based compounds in its molecular structure. A share of these compounds is released as CO_2 when the lime is applied. The emission factor portraying the share of carbon released is obtained from IPCC (2019a). A conversion factor of 44/12 is used to transform the carbon emissions into CO_2, which is obtained from the same source.

The dataset provided by Brentrup et al. (2018) discussed in chapter 3.5.4 is utilized to account for the emissions from the manufacture and industrial preparation of the lime.

3.5.3 Urea application

Urea and urea-based fertilizer contain carbon in their structure. This carbon is fixated in the molecular structure of the fertilizer during its manufacture and is released when the fertilizer is applied to the soil. The emission factor for the carbon emissions and the conversion factor of 44/12 to transform them into CO_2 are provided by IPCC (2019a). For urea-based fertilizers, the emission factor is applied only to the share of urea contained in the fertilizer.

3.5.4 Manufacture of fertilizer

The GHG emissions from the manufacture of fertilizer depend, among other things, on the type of fertilizer as well as on the source of the electricity used in its production. Different raw materials may be required to produce diverse fertilizers. These must be extracted and transported to the factory to be processed. The chemical reactions from the processing of the raw materials also release GHG emissions. The intensity of these gases depends on the technologies used in the factories (Brentrup et al., 2018).

The emissions from the generation of electricity also are considered in the carbon footprint of the fertilizer. Thus, fertilizers manufactured in regions that have a higher share of electricity from renewable sources have a comparatively lower carbon footprint. Moreover, the GHG emissions from electricity generation vary substantially depending on the source of the fuel used as well as the technology utilized in the power plant (Zhang et al., 2013). Consequently, the same fertilizer can have considerably different carbon footprints depending on where it is manufactured.

Brentrup et al. (2018) provide a dataset with the emission factors from the manufacture of fertilizers, disaggregated fertilizer and region of the world, which is used in this analysis. The dataset is an expansion of the assessment by Hoxha and Christensen (2018). These values account for the GHG emissions from the extraction and transportation of the raw materials as well as from their processing, including the carbon footprint from electricity generation in the region. Moreover, the values already include the carbon sequestered in urea and urea-based fertilizers occurring during the manufacture (see chapter 3.5.3). For this calculation, it is assumed that all the fertilizers the grower uses are produced in the same country where the Typical Farm is located.

3.5.5 Diesel manufacture and use

The combustion of diesel[6] releases approximately 2.6 kg CO_{2eq}/l (Robertson and Grace, 2004). No dataset providing region-specific emission values for the extraction of crude oil and refining it into diesel could be identified. Therefore, a global average is used instead. Masnadi et al. (2018) calculated a global weighted average and indicate that the emissions released from the extraction and transport of crude oil to the refineries are 10.3 g CO_{2eq}/MJ. Refining the crude oil into diesel and distributing it releases an additional

[6] Robertson and Grace (2004) indicates that 706 g of carbon are released per liter of diesel. To transform from carbon to CO_2, a factor of 44/12 is utilized, which is obtained from IPCC (2019a).

5.3 g CO_{2eq}/MJ (Atris, 2020). The energetic content of diesel is assumed to be 35.3 MJ/l (Smallbone et al., 2020). Hence, the extraction, transport and refining to obtain diesel releases 0.6 kg CO_{2eq}/l. This value is added to the emissions from the combustion. Hence, diesel usage on all Typical Farm implies a release of 3.1 kg CO_{2eq}/l.

3.5.6 Land use change

Land use change depicts the GHG emissions from variations in the use or management of the soil. These result from changes in the soil carbon content. A reduction in the content implies its release as CO_2. Contrarily, an increase signifies carbon sequestration.

IPCC (2019a) methodology is used to calculate the emissions from land use. This methodology is based on the supposition that soil organic carbon reaches a stable stage (or equilibrium) after sufficient time of constant soil management has passed since the last change to the system occurred. Hence, no net change in the carbon stock takes place.

The main soil category must be defined to apply the methodology. Typical Farms are assumed to operate on mineral soils unless scientific evidence or the focus group indicate otherwise[7]. The total carbon content of mineral soils in equilibrium is calculated using Equation 21.

$$SOC_{total} = SOC_{ref} \times Coef_{use} \times Coef_{tillage} \times Coef_{input} \qquad (21)$$

SOC_{total}:	mineral soil organic carbon stock in the system in equilibrium (t C/ha)
$SOC_{reference}$	mineral soil organic carbon in reference soil (t C/ha)
$Coef_{use}$:	coefficient for land use (annual crop, perennial crop, rice, etc.)
$Coef_{tillage}$:	coefficient for tillage intensity (full, reduced, no-till)
$Coef_{input}$:	coefficient for input of organic matter (low, medium, high with or without manure)

IPCC (2019a) provides all the coefficients necessary for this calculation, but these are used only if no region-specific data can be found in scientific literature. Soil organic carbon in reference soil (SOC_{ref}) can be interpreted as the carbon stock before human intervention and is based on the soil type and climate. Likewise, the coefficients depicting land use, tillage intensity and input of organic matter are provided, disaggregated by climate regime. A definition of each element and its categories is provided in chapter 3.6.2.2 for tillage intensity and chapter 3.6.2.3 for input of organic matter.

Equation 21 is used to calculate the resulting carbon stock under the mitigation scenario, which is achieved by replacing the corresponding coefficient; e.g., from full tillage to reduced tillage. The difference between the carbon stocks in both scenarios represents the net change in emissions. This difference in stocks is

[7] Mineral soils differentiate from organic soils principally based on the organic carbon content of the soil. In general terms, soils with less than 12% are considered mineral, according to IPCC (2019a).

divided by the time it takes the soil to transition to its new equilibrium to annualize the calculation (Equation 22).

$$CO_2 - LUC = \left(\frac{SOC_{total(baseline)} - SOC_{total(strategy)}}{Transition\ time}\right) \times CF_C \qquad (22)$$

CO_2-LUC:	annual GHG emissions from land use change (C/a)
$SOC_{total(baseline)}$	mineral soil organic carbon in baseline scenario (t C/ha)
$SOC_{total(strategy)}$:	mineral soil organic carbon with mitigation strategy (t C/ha)
Transition time:	transition period between original and new equilibrium (years)
CF_c:	conversion factor from carbon to carbon dioxide (equal to 44/12)

According to IPCC (2019a), the transition time can be assumed to be 20 years. It can be inferred that the methodology assumes that the change transition between carbon stocks occurs linearly. Moreover, the methodology also suggests that, to maintain the carbon sequestered by the strategy, the changes that generated the net increase in carbon must be used permanently. If the practices no longer are conducted, the carbon sequestered is released as emissions; i.e., carbon sequestration is reversible.

Hence, in the case of strategies that promote carbon sequestration, two different stages are assumed. The transition stage of 20 years denotes the time in which a net change in the carbon content of the soil results from the mitigation strategy; i.e., carbon sequestration takes place. This is followed by a stable stage, that represents the carbon content of the soil with the strategy has reached its equilibrium. Thus, the carbon sequestration potential is realized and becomes zero. The implications and possible adjustments between the stages are discussed in the corresponding chapters. The changes in soil organic carbon assumed in the case studies are presented in the Appendix (Table A.14 for the case study in the USA [Iowa] and Table A.15 for Germany [Mecklenburg-Vorpommern]).

3.5.7 Summary of emission categories

The elements entailed in each GHG emissions category presented for the mitigation strategies are:

(1) **Direct** depicts the N_2O emissions released by the chemical transformation of the nitrogen cycle occurring in the soil. It is driven by the use of nitrogen fertilizer and the decomposition of crop residues.

(2) **Indirect** represents the N_2O emissions caused by leaching and volatilization of nitrogen, which are modified using the adjustment factor. The drivers of this category are the same as for direct.

(3) **Liming** entails the GHG emissions from the extraction and processing of lime as well as the CO_2 released by its use.

(4) **Urea** entails the CO_2 emissions from the usage of urea, which results from its chemical composition. The N_2O emissions triggered by nitrogen in the urea are accounted for in the categories direct and indirect.

(5) **Manufacture** portrays the GHG emissions from the manufacture of the fertilizers.

(6) **Diesel** represents the CO_2 emissions from extraction, refinement and use of diesel for machinery.

(7) **Land use** represents the soil carbon emissions or sequestration occurring from changes in the management of the soil.

3.6 Mitigation strategies assessed

The selection of the mitigation strategies evaluated for the Typical Farms in the case studies was conducted in two stages: compilation of strategies and evaluation of the feasibility to generate representative results.

Compilation of strategies in scientific literature

The first stage was the compilation of a list of strategies usually discussed in scientific literature. Studies at farm-level from the regions and crops selected for this project were reviewed first, followed by strategies not previously assessed or from other regions. In this stage, these are evaluated on the fundamental principle proposed to mitigate emissions rather than the technical implications. The strategies identified in the literature[8] can be divided into whether they focus on reducing nitrogen and fertilizer-driven emissions or promote carbon sequestration by increasing soil organic carbon.

(1) Reduction of nitrogen and fertilizer-driven emissions:

a. **Optimization of nitrogen fertilization rate:** The nitrogen rate potentially can be lowered if the timing of the application is adjusted to better match the requirements of the crop or the number of passes to spread fertilizer is increased. Precision farming tools can also be utilized to lower the rate based on the availability of the nutrient.

b. **Utilization of organic fertilizer:** Manure is discharged into the environment in some parts of the world or is used inefficiently as fertilizer. This leads to the loss or accumulation of nutrients that could have been used to fertilize other fields. Improving the usage efficiency of manure would enable a reduction in the rate of nitrogen and potentially other types of fertilizer as well.

c. **Legumes as cover crops:** Legumes can be used as cover crops before the cash crop is established. Legumes biologically fixate nitrogen that can become available to the following crop, which enables a reduction in fertilizer rate. Moreover, the crop residues of the cover crops can be incorporated into the soil, promoting carbon sequestration. Moreover, cover crops have the potential to reduce nutrient losses by taking them up in their structure. These are released when the residues decompose. The nutrients become available to the next crop, lowering the rate of fertilizer needed.

d. **Enhanced efficiency fertilizers:** Chemical additives such as inhibitors can be used with nitrogen fertilizer. This affects the transformation rate of the fertilizer in the soil, reducing the losses as gas or via leaching and volatilization. As a result, a greater proportion of the nutrient is potentially available to the plant, which means that less fertilizer can be applied to provide the same amount as without the additive.

e. **Adjustment of the fertilizer bundle:** Nitrogen fertilizers have different carbon footprints from their manufacture. The bundle of fertilizer used can be replaced by alternatives that entail reduced emissions per unit of nutrient while maintaining the total rate constant.

(2) Carbon sequestration:

a. **Reduction of tillage intensity:** The disturbance of the soil via tillage promotes the release of the carbon contained in the organic matter. Thus, reducing tillage intensity by using shallower soil management practices or equipment that reduces the share of the soil disturbed can promote carbon sequestration.

b. **Stopping crop residue burning:** Residue burning releases N_2O and implies the foregone opportunity to incorporate the crop residues into the soil. Hence, replacing residue burning with

---

[8] The literature used for the elaboration of the list of possible mitigation strategies is: Feliciano et al. (2018); Smith et al. (2008); Schneider et al. (2007); MacLeod et al. (2015); Wolt (2004); Lal (2004b); Akiyama et al. (2010); Noble and Christmas (2008); Pellerin et al. (2017); Cerri et al. (2010); Smith et al. (2013); Osterburg et al. (2019); Osterburg et al. (2013); Petersen et al. (2013); Pérez Domínguez et al. (2020).

alternative practices that leave the residues on the soil or incorporate them results in carbon capture.

c. **Cover crops:** the residues from crops can be perceived as an additional source of biomass. This promotes an increase in soil organic carbon if incorporated into the soil.

Feasibility to generate representative results in the Typical Farm

The second stage is the evaluation of the feasibility to generate representative results in the context of the Typical Farms. For this purpose, the complete list of strategies is evaluated in each case study using the same considerations:

(1) **Strategy is part of baseline**: This initial evaluation determines whether or not the strategy can be assumed to be a part of the baseline scenario; e.g., reducing tillage intensity cannot be implemented if no-till is already the status quo. This aspect is assessed using statistical data and selected interviews, and is discussed in the focus group.

(2) **Participants of the focus group are acquainted with the strategy:** the next consideration is that the participants of the focus group and the scientific partner must have a basic understanding of the strategy and its implications at the farm level for it to be evaluated properly. This is necessary to correctly assess the strategy's challenges; for instance, the availability of inputs or requirements of machinery. This aspect is considered key to deriving realistic and applicable results.

(3) **Availability of scientific data:** Additionally, scientific literature, preferably from the region of the case study, must be available to validate the assumptions necessary for the calculation of the mitigation strategies. Alternatively, data from regions with comparable conditions will be used. Yet, if not enough applicable data are found, the strategy is not included, as it is deemed that it cannot be properly assessed.

(4) **Risk of net increase in emissions from reduced yields:** A literature review to determine possible impacts on crop yields resulting from the mitigation strategy is conducted, which is complementary to the discussion in the focus group. A permanent yield decrease without a respective reduction in demand would lower global food supply and would entail additional land to be transformed for agricultural use and/or the existing production would need to be intensified, implying emission leakage (see chapter 2.4). Taking into account this problem, strategies that result in unavoidable yield reductions are not considered feasible if data is insufficient to assess the risks of ILUC. The intent is to prevent a net increase in global emissions from occurring. Nonetheless, in cases in which it can be estimated that the emission reduction from the strategy on the Typical Farm is larger than the emissions from ILUC, the strategy is still evaluated in this thesis.

Only the strategies that were effectively assessed in this thesis - that is, they meet these criteria - are presented and discussed further. The arguments that contributed to the decision not to evaluate the remaining strategies are shown in the Appendix in Table A.3.

Combination of mitigation strategies

The possibility to combine all the strategies evaluated for a Typical Farm is discussed in this thesis. This scenario is constructed from the assumptions and results of the individual mitigation strategies. Consequently, this strategy is not included in the focus group discussion. Nonetheless, this is not considered to significantly limit the feasibility or the comprehensiveness of the combination of strategies, as the individual elements are thoroughly discussed in the focus group.

3.6.1 Mitigation potential and explanation of cases

The formulas used to estimate the mitigation potential are obtained from the literature revised for compilation of the list of mitigation strategies. Some of the studies reviewed utilized the methodology provided by IPCC (2019a) as presented in chapter 3.5 for this purpose. However, in other cases, additional calculations are proposed. These calculations are a supplementary methodology applied only for the calculation of the specific mitigation strategy. The methodology and its sources as assumed for each strategy in this thesis are explained in the following respective chapters.

In addition, the implementation of these methodologies requires coefficients for the calculation. While some are provided in the guidelines from IPCC (2019a), others must be retrieved from scientific literature. In general, the coefficients portray information that is specific to a crop or region - for instance, nitrogen supply from biological fixation by legumes as cover crops. Moreover, these coefficients usually are provided as a range. This range can be interpreted as the variation in the mitigation potential that can be attained with a specific strategy. To illustrate, the nitrogen fixated by a legume determines the potential to lower the nitrogen fertilizer rate, which defines the emission reduction. In this case, the highest fixation rate from the range implies the highest reduction in GHG emissions. Thus, in this thesis, if ranges are provided, they are used to construct three cases to display the variation in the mitigation potential: standard, best, and worst.

The standard case represents the average mitigation potential that can be expected with the mitigation strategy. This is based on the average coefficient as defined by the authors of the study from which the coefficients are obtained. The best case is constructed using the values from the range that result in the highest mitigation potential. The worst case is derived from the values depicting the lowest possible mitigation potential.

It must be noted that defining the three cases with this approach limits their comparability. This is because the ranges reported in the literature and implemented in this thesis may have been determined using different approaches that may not be comparable or relevant statistical data may not be reported. Some variables utilized in the GHG calculation are provided with a statistical distribution; for instance, the values from IPCC (2019a), which are implemented in this assessment. However, other variables from scientific literature used in the assessment of the individual strategies are indicated as ranges or unique values without further clarifications on the statistical description of the data. In these cases, especially if empirical data is missing, assuming a statistical distribution increases the uncertainty of the emission and cost calculations (Lee et al., 2017). This limits the possibility to compare the three cases as it is not possible to systematically assign probabilities to each variable. Consequently, it is problematic to determine the resulting statistical deviation from the true mean of each mitigation strategy (Lee et al., 2017). As a result, the cases, as defined in this thesis, indicate only the range of mitigation costs the strategy offers in the Typical Farm. Nonetheless, the mitigation costs calculated for each case of a strategy as well as across strategies are compared qualitatively at the of each case study.

3.6.2 Strategies evaluated in the Typical Farms

The general considerations and implications at farm level resulting from the adoption of the strategies are presented in this chapter. These considerations help understand the decision-making of the participants in the focus group. Only the strategies that were evaluated based on the criteria previously presented are covered. The methodology used to estimate their mitigation potential and cases are explained.

3.6.2.1 Optimization of nitrogen rates

Nitrogen is one of the most important nutrients and limiting factors for the crop's growth (Kraiser et al., 2011). Crops take nitrogen up from the soil and utilize it for various metabolic processes. Without enough of it, they are not able to grow at their potential, which, in turn, reduces yields. At the farm level, lower yields negatively affect revenue[9]. Furthermore, higher yields imply that less land is required to supply sufficient production to satisfy the demand. Therefore, fertilizing with nitrogen is necessary to be environmentally and economically sustainable (McAllister et al., 2012), which implies that GHG emissions from nitrogen fertilization are partly unavoidable.

Equivalent to an undersupply of nitrogen, an oversupply has negative consequences to the environment and the economic performance of the crop, as it increases losses. The rate at which nitrogen will be lost is determined by the strength of the bounds the soil particle forms with the nutrient (Sollins et al. 1988). The nutrients arrange themselves in layers around the soil particles, with the outer layers being bound by weaker links compared with the inner layers .This indicates that the closer the soil is to its saturation point, the likelier it is to lose the nutrient to the environment. Consequently, the more fertilizer applied in a single pass, the higher the amount of nitrogen that can be lost as well as increased odds it will be lost (Perakis et al., 2005), ultimately increasing costs. Therefore, to minimize these costs, growers would theoretically split their nitrogen applications into as many passes as possible.

However, each pass to apply fertilizer costs diesel and labor, and locks out the possibility to use those resources in other operations. These total application costs are affected only minimally by the amount of fertilizer applied per hectare, which is a function of the output rate of the spreading machine and the speed at which it is conducted. The output speed of the fertilizer spreader can be adjusted to various rates to compensate for higher or lower speeds. This enables the grower to decide the speed of the pass on the field. As with cars, a tractor's engine has a specific range in which its fuel efficiency is the highest, which determines the tractor's speed. The speed also determines labor costs for each pass since these are based on the time that it takes to complete the operation.

The utilization of the machine also generates opportunity costs since it limits the possibility to use it for other operations. However, to lower the opportunity costs, the grower would need to increase the speed, which would increase the fuel costs. Hence, applying fertilizer involves costs that can be minimized regardless of the amount of fertilizer being applied. The grower can perceive them as the fixed costs of each pass. Consequently, the grower will try to reduce the number of passes to the minimum.

Yet, reducing the number of passes implies that higher rates of nitrogen must be used per application. This increases the amount of nitrogen that can be lost to the environment, increasing costs and pollution. In an ideal case, the grower knows how much nitrogen is taken up by the plant and how much is lost. Thus, to minimize costs, the grower will be inclined to increase the number of passes if the cost of an additional pass is lower than the nitrogen losses prevented by the additional pass. Still, it is likely that the nitrogen rate from this economically optimized fertilization plan is larger than what is required to maintain the yields, as covering the cost of the lost nitrogen is cheaper than an additional pass. Thus, unnecessary N_2O emissions are released since more nitrogen is applied than required by the system. In such cases, it is possible to mitigate GHG by increasing the number of passes, as this would enable the grower to lower his nitrogen application rate without negatively affecting yield.

Moreover, growers have the choice to implement precision-farming techniques such as variable-rate application to increase the efficiency of nitrogen fertilization. For instance, this system could be

[9] To illustrate the importance of nitrogen in the economic performance of crops, each dollar that the German Typical Farms invested in nitrogen fertilizer when growing wheat produced an economic return of 11.53 USD (average values from 2016 to 2019 across all farms).

synchronized with historic yield maps or nitrogen-sensing technology. These types of technologies have been widely adopted by farmers in some regions of the world, such as in the USA (Schimmelpfennig, 2016). Nonetheless, these systems are not available everywhere and their acceptance among producers is generally low (OECD, 2016). These systems imply additional expenditures that may have low economic returns (Liu et al., 2006; Gandorfer and Meyer-Aurich, 2017), entail a significant uncertainty regarding the response of the yield (Morris et al., 2018) and require further know-how. These elements can become significant barriers to adopting these types of technologies (Paustian and Theuvsen, 2017; Roberts et al., 2004).

Nevertheless, growers do not usually have precise information on the amount of nitrogen taken up by the crop and lost to the environment. More importantly, it is not feasible to accurately predict the expected losses that will occur in the remainder of the growing season. The estimation of the crop requirements depends on the grower's know-how regarding the absorption rates of the plant and the effect that the production strategy has in the system. The amount lost to the environment depends on aspects that are uncertain at the moment of deciding the fertilization rate, such as weather conditions during the growing season and the yield (Spangler et al., 2017). Further aspects such as the nitrogen left in the soil from the previous growing season can be measured with soil sampling or indirectly by sensing the nitrogen content in the plant to provide the grower with additional information. Still, given the importance of the nutrient for the crop's development and the profitability of nitrogen applied, growers are likely to utilize more than what they estimate the crop needs to cover for unexpected losses.

If this is the case, it would be theoretically possible to mitigate GHG emissions by reducing the fertilizer rate to cover only the requirement of the crop to attain the expected yield and the soil microorganisms, eliminating the excess nitrogen (see chapter 2.2). This reduction could be achieved without additional passes. By maintaining the same number of passes but reducing the nitrogen rate to the required level, GHG emissions could be reduced, without changing yields. This strategy potentially could reduce costs for the grower and reduce GHGs from the system, generating a win-win situation.

It also is possible the grower is underestimating the nitrogen his system requires. In this case, increasing nitrogen rates could indirectly reduce emissions. As explained earlier in this chapter, not using enough nitrogen depletes the reserves of nutrients in the soil. This reduces the fertility of the soil in the long term and decreases yields. Considering that the demand for the crop remains unchanged, more land would need to be converted to arable land, releasing GHG from the associated land use change. Consequently, increasing the nitrogen rate would increase the emissions from the field, but would avoid the release of additional emissions from ILUC. Additionally, the higher rate would increase yields, improving the income of the grower.

Estimation of the reduction potential and cases

Research papers and GHG estimation tools were revised to determine which alternatives are available to calculate the potential reduction in nitrogen rates. It was revealed that most of the recent literature and tools utilize the Nitrogen Use Efficiency (NUE) methodology developed by Brentrup and Palliere (2010). To make the results of this project comparable with other research, this approach therefore was adopted in this project.

The NUE suggested by Brentrup and Palliere (2010) is based on the calculation of the proportion of nitrogen that is removed from the system through harvesting of crop parts compared with the total nitrogen supplied

(Equation 23). The calculation of nitrogen supply and nitrogen removed via harvest is presented in chapter 3.5.1.2.

$$\boldsymbol{NUE_{baseline}} = \frac{\boldsymbol{N_{harvest}}}{\boldsymbol{N_{supply}}} \times 100 \qquad (23)$$

NUE: nitrogen-use efficiency (%)
$N_{harvest}$: amount of nitrogen exported from the system via harvest (kg N/ha)
N_{supply}: sum of nitrogen inflows (fertilizer and biological fixation) available to the crop (kg N/ha)

The NUE calculated in the baseline scenario is compared with the optimum NUE for the yield obtained in the Typical Farm. Brentrup et al. (2004) provide NUE values that can be used as a reference for optimum nitrogen fertilization rate. Achieved without negatively affecting yield, Brentrup and Palliere (2010) mention that the optimum NUE is between 80% and 90%. They further argue that removing 100% of the nitrogen supplied is not sustainable because, as explained earlier, soil bacteria also need to be supplied with nitrogen to avoid soil mining and a part of the nitrogen applied will be naturally lost to the environment. Hence, the 10%-20% necessary to reach 100% NUE represents the share provided to the microorganisms and covers the natural losses. However, these depend on the bacteria composition, soil and climate conditions, meaning that an optimum NUE is specific to each location. For each case study, a literature review is conducted to use location-specific optimum NUE values. If no values are identified, Brentrup et al. (2004) values will be used, in which case, a NUE of 85% is defined as optimum in the standard case, with 90% and 80% as the best and worst cases. The optimum supply of nitrogen in each case is calculated with Equation 24.

$$\boldsymbol{N_{optimum\ supply}} = \frac{\boldsymbol{N_{harvest}}}{\boldsymbol{NUE_{optimum}}} \qquad (24)$$

$N_{optimum\ supply}$: optimum supply of nitrogen to obtain the yield expected in the Typical Farm (%)
$NUE_{optimumt}$: optimum NUE based on literature (%)

The difference between the nitrogen supply in the baseline scenario and the optimum calculated for each case indicates the nitrogen reduction potential achievable with the strategy (Equation 25). A preliminary estimation of the reduction in the fertilizer rate is discussed in the focus group to determine which specific fertilizer is reduced, in case more than one is applied.

$$\boldsymbol{N_{reduction\ potential-NUE}} = \boldsymbol{N_{supply}} - \boldsymbol{N_{optimum\ supply}} \qquad (25)$$

$N_{reduction\ potential\text{-}NUE}$: nitrogen reduction potential from NUE (kg N/ha)

Theoretically, the nitrogen reduction potential calculated indicates how much nitrogen can be subtracted from the fertilizer rate without negatively affecting yield, since the approach minimizes the losses. Nonetheless, a literature review is conducted in each case to validate this assumption.

3.6.2.2 Reduction of tillage intensity

Tillage accomplishes several functions in an arable system. Primarily, this operation is used to prepare the soil to facilitate seeding and development of the crop. Tillage disturbs the soil by breaking, cutting, mixing or inverting it (Cannell, 1985). Several layers of soil can be affected by this operation. These are redistributed, promoting the oxidation of soil organic carbon (Shepherd et al., 2001). As a result, carbon is released as CO_2. The carbon content replenishes via the decomposition of the crop residues since a share of the carbon contained in the plant's structures can be incorporated into the soil. Hence, it can be inferred that, if the input of carbon from crop residues is held constant and the loss as emissions is reduced, the soil organic carbon would increase, indicating carbon sequestration. In this regard, decreasing the disturbance of the soil by reducing tillage intensity can reduce carbon losses. Decreasing tillage intensity can be understood as utilizing equipment that operates at a shallower depth, leaves a larger portion of the soil undisturbed, or both.

Carbon sequestration via reduced tillage, especially no-till, has been questioned in the scientific community (Nicoloso and Rice, 2021; Ogle et al., 2012). Early meta-studies suggested that the potential carbon sequestration was considerable (West and Post, 2002; Ogle et al., 2005). However, later reviews indicated that the total carbon content of the soil is not increased if the depth of the soil analyzed is increased (Angers and Eriksen-Hamel, 2008; Luo et al., 2010; Olson et al., 2014). The authors argue that a reduction of tillage leads to a redistribution of the carbon, which accumulates mainly in the upper layers, but the net amount remains unchanged. Nonetheless, current reviews have accounted for more variables and have utilized larger datasets. They have found that adopting no-till can effectively promote carbon sequestration, although the potential is indicated to be lower than previous findings suggested (Nicoloso and Rice, 2021; Bai et al., 2019), although the effect may be negligible under certain conditions (Jacobs et al., 2018). The effects can differ considerably based on the soil and region considered. Nonetheless, this complex topic cannot be properly evaluated in this thesis and the prevailing data and approach obtained from IPCC are utilized. In this regard, these new insights are reflected in the guidelines from IPCC. A comparison of the coefficients for no-till from the 2006 guidelines with the coefficients from 2019 reveals a reduction in the carbon sequestration potential of up to 70% for selected regions.

Reducing tillage intensity can have negative implications in crop production. The soil is compacted each season by interactions specific to the crop, soil, climate and, especially, by agricultural machines (Soane, 1994). A reduction in tillage intensity entails that these compacted portions are broken up to a lesser extent and, as a result, the soil usually is less loose (Ohiri and Ezumah, 1990). This can cause problems, especially in establishing the crop. Seeding equipment may not be able to operate efficiently in compacted soils, generating uneven crop establishment and development, likely hampering yields (Lavoie et al., 1991). Moreover, the germination of the seeds may be depressed by compaction, also likely lowering yields (Hyatt et al., 2007). Furthermore, tillage suppresses weed populations. Hence, reducing tillage intensity could imply the need to increase the application of herbicides (Cannell, 1985; Meade and Mullins, 2005). Consequently, reducing tillage intensity can entail considerable challenges for the grower.

Nonetheless, reducing tillage intensity also can positively affect yields. Lowering the disturbance of the soil reduces the risk of erosion, can improve soil structure in the long-term, and increases the water-holding capacity of the soil (Holland, 2004). Particularly this last aspect can be advantageous in conditions of limited water supply. The increase in the soil's organic carbon content also can increase soil fertility, leading to higher yields and more efficient utilization of inputs (Haddaway et al., 2017).

Furthermore, tillage equipment is specialized for the type of operation it accomplishes; hence, using it for other purposes is not feasible. Reducing tillage intensity can result in the grower no longer needing to maintain this equipment in its inventory, lowering machinery costs. Tillage, in many cases, is the operation that demands the greatest power of the tractor. Thus, decreasing tillage intensity, in certain cases, can enable the farmer to utilize tractors with comparatively less power, further lowering costs. However, seeding equipment may need to be adapted to the new system, which can imply a replacement of the equipment or parts of it (Rainbow and Derpsch, 2011).

Shallower operations decrease fuel and labor expenditures as the operations can be conducted in a shorter time, further lowering costs. Moreover, the decrease in fuel consumption lowers the emissions from the system.

Nonetheless, it can be argued that adjusting tillage management entails considerable changes to the farm's operations and equipment. In this regard, a significant barrier to transitioning to a lower tillage intensity can be the grower's know-how (Jayaraman et al., 2021).

The improvement in soil structure and increase in soil fertility occur gradually over extended periods (Roger-Estrade et al., 2009). While these changes occur, yields can be comparatively lower due to problems with compaction. Thus, the farm's profitability may decrease for multiple seasons before yields recover. Consequently, the process of transitioning to a reduced tillage intensity is linked to several crop and region-specific factors the grower must consider.

Estimation of the reduction potential and cases

The methodology from IPCC (2019a) presented in chapter 3.5.6 and often used in research papers is used to estimate the carbon sequestration potential of reducing tillage intensity.. Following the methodology, tillage intensity is depicted by a coefficient ($Coef_{tillage}$) containing three alternatives. In this thesis, the definitions of these alternatives are based on the guidelines as well as the descriptions by Gebhardt et al. (1985).

(1) **Full tillage:** Entails total or partial soil inversion - for instance, via ploughing - as well as the use of cultivators. Less than 30% of the soil is covered with residues at the time of planting the new crop.

(2) **Reduced tillage:** No soil inversion occurs. Cultivators and other equipment for shallow operations can be used to incorporate the residues in the topsoil soil. At least 30% of the soil is covered with residues from last season.

(3) **No-till:** No tillage operation occurs and 100% of the residues are left on the surface. Crop is seeded directly through the residues from last season.

According to the methodology from IPCC (2019a), each tillage intensity from the list corresponds with a coefficient within a range, which are provided in their methodology. The baseline scenario is calculated using the average from the range. Similarly, for the mitigation strategy, the standard case is defined using the average of the range for the tillage intensity defined. The upper value of the range defines the best case, as this implies the highest increase in soil carbon stock, which implies the highest carbon sequestration potential. Contrarily, the worst case is constructed using the bottom value from the range because this depicts the lowest mitigation potential.

The tillage management of the Typical Farm in the baseline scenario is defined in the focus group. The discussion includes the specific operations to corroborate the allocation based on these definitions. If the mitigation strategy is feasible, whether the tillage can be changed to reduced tillage or no-till, is determined with the same approach. In this regard, the machinery affected by the change is characterized. Moreover, in each case study, a literature review is conducted to validate the feasibility of promoting carbon sequestration with this approach in the selected region. Likewise, this is also conducted to evaluate possible impacts on yields.

3.6.2.3 Cover crops and legumes as cover crops

Carbon sequestration in the soil also can be promoted by increasing the input of carbon. Winter cover crops usually are recommended as an additional source of biomass to the system that can replace winter fallow (Mazzoncini et al., 2011). The residues can be incorporated into the soil to promote its decomposition and increase the carbon stock (Poeplau and Don, 2015; Jian et al., 2020; Bai et al., 2019). In the long term, this improves soil structure, increasing yield stability (Lotter et al., 2003). Additionally, cover crops promote biodiversity and protect land against erosion (Lal, 2004b) as well as suppress weeds (Snapp et al., 2005).

Moreover, cover crops can reduce leaching, decreasing nutrient losses – especially nitrogen - from the soil (Blombäck et al., 2003). The nitrogen is absorbed into the plant's structure and mineralizes, becoming available when the residues decompose. Thus, this nitrogen can be utilized by the following crop, reducing the rate of synthetic fertilizer needed (Kuo and Sainju, 1998; Snapp et al., 2005). Additionally, legumes can be used as cover crops. They biologically fixate nitrogen, further increasing the supply of the nutrient for the following cash crop (Liu et al., 2011).

However, the added crop residues can negatively affect yields of the next crop because they decrease soil temperature by blocking sunlight (Tonitto et al., 2006; Stivers-Young and Tucker, 1999). This risk can be reduced by tilling the residues, yet this operation can be considered additional as it may not be a part of the baseline. Growers may spray herbicide to terminate the cover crop to reduce the risk of interfering with the cash crop or becoming a permanent weed in the system (Benech-Arnold et al., 2000), increasing the expenditure on pesticides. The establishment of the cover crop implies additional seed and seeding operation costs. All these operations imply added machinery, labor and fuel costs, as well as emissions from diesel consumption, and may compete with other operations. Hence, using cover crops can increase costs considerably.

The selection of the cover crop is critical to ensure a successful establishment while minimizing negative impacts on the cash crop. Cover crops that do not share pests or diseases with the cash crops should be used. The species should be able to develop sufficient biomass to cover the soil in the time it has to develop. Climate is particularly relevant in this selection as species prone to die from low temperatures - i.e., suffer winterkill - can be advantageous because they would not need to be terminated with herbicides. Moreover, a mixture of plants with complementary characteristics can be used instead of a single species. Legumes can fixate nitrogen, but their seeds are comparatively more expensive and have a limited potential to generate biomass (Snapp et al., 2005). The equipment of the grower is also a deciding factor, as it may not include a seeder compatible with the cover crop. Similarly, the farmer may not know how to manage cover crops, or the specific species adopted. It can be inferred that the decision of which cover crop to use is governed by a series of complex factors and interactions.

Estimation of the reduction potential and cases

Cover crops offer two possibilities to mitigate emissions that can occur simultaneously: carbon sequestration and a reduction of the nitrogen rate. The methodology shown in chapter 3.5.6 obtained from IPCC (2019a) is used to calculate the carbon sequestration potential. The methodology uses coefficients ($Coef_{input}$) depicting the input of biomass to the soil. Four different alternatives are presented, adapted from the guidelines:

(1) **Low:** Residues from annual crops are removed from the field or burned. No synthetic fertilizer is applied.

(2) **Medium:** Residues from annual crops are left on the field or incorporated. Synthetic fertilizer is applied.

(3) **High without manure:** Same as medium but with an additional input of biomass via cover crops or irrigation of the cash crops.

(4) **High with manure:** Same as high without manure but with a regular application of manure.

The definition of the standard, best, and worst cases is based on the ranges provided by IPCC (2019a) for each coefficient, as already explained in chapter 3.6.2.2.

The mitigation potential from a reduction of the nitrogen rate is the result of two possibilities: the prevented losses as leakage and the biological fixation from legumes. Levels of nitrogen the cover crop absorbs and makes available to the following crop as well as the biological fixation by the legume are obtained from scientific literature. The three cases are based on the ranges provided in the studies: The standard case is the average of the range; the best case implies the highest potential nitrogen prevented from leaching or fixated by the legume; and the worst case, the lowest prevention or fixation potential. The nitrogen provided to the cash crop from either possibility is reduced from the synthetic fertilizer rate, which is selected in the focus group.

The adjustment factor (see chapter 3.5.1.2) is not affected by the reduction in the nitrogen fertilizer rate. Nitrogen from the avoided leaching and biological fixation is assumed to replace the fertilizer, indicating that the total nitrogen input remains unchanged.

3.6.2.4 Enhanced-efficiency nitrogen fertilizers

Enhanced-efficiency fertilizers can be understood as nitrogen fertilizers treated with additives to control the speed of release of the nutrient or to modify the reactions that occur in the soil (Halvorson et al., 2014). According to Akiyama et al. (2010), there are three main types of additives that differ in their function: nitrification inhibitors, urease inhibitors and slow-release coating. There are several alternatives within each additive but they all pursue the same objective, which is to increase the availability of nutrients to the crop by reducing losses.

The use of enhanced-efficiency nitrogen fertilizer can decrease nitrogen losses from various pathways. In general, they lower the losses via leaching, emissions as N_2O and volatilization as NH_3 (Akiyama et al., 2010; Li et al., 2018; Thapa et al., 2016; Wolt, 2000). The effect on each source of losses depends on the type of enhanced fertilizer and on the specific formulation of the additive. Thus, two different urease inhibitors may alter nitrogen losses differently. It also is possible to combine several additives of the same or different types. Factors such as the form of nitrogen contained in the fertilizer, crop, soil and climate are equally relevant in the effect of the additive on losses (Thapa et al., 2016; Li et al., 2018).

In principle, additives reduce losses by affecting the transformation rate of nitrogen in the soil, which could be understood as a direct increase in the supply for the crop. Hence, if the supply of fertilizer is not sufficient, the additives can increase yields by increasing availability. However, decreasing the transformation rate may have the opposite effect, limiting the timely supply for the plant and hampering the crop’s development. Nonetheless, the exact effect on yields depends on multiple factors.

Overall, enhanced-efficiency fertilizers are not usually used by growers as they rarely observe an improvement in the yields (Guertal, 2009; Timilsena et al., 2015). Hence, although their adoption on the farm does not require considerable technical adjustments as it is likely that the existing equipment and operations can be maintained, the additives are perceived as an unnecessary cost.

Depending on the circumstances, the use of enhanced-efficiency fertilizer can increase pollution. Assume that the nitrogen applied is either exported as harvest or lost to the environment. If the additives result in lower yields, the nitrogen that no longer is exported due to the reduced yield must be lost. This signifies comparatively more leaching or emissions. Alternatively, the additives may not have any significant effect on the system overall, suggesting that the losses and yields are the same as without them. If yields are not affected, the additives could generate pollution swapping (Drury et al., 2017) - a term that describes an increase in the nitrogen losses in one pathway as a result of mitigation of another one (Stevens and Quinton, 2009). For instance, assuming yields are unchanged, the nitrogen that no longer is lost as leaching because of an inhibitor must be lost via volatilization or emissions. Consequently, it can be assumed that if no change in yields occurs, the utilization of the additives should be paired with a reduction in the nitrogen rate (Li et al., 2018). The reduced rate generates the need for the additional supply derived from the additives.

Estimation of the reduction potential and cases

No applicable approach that included a comprehensive integration of the changes in the losses via emissions, leaching and volatilization as well as in yields could be identified in literature. Therefore, the approach used in this thesis is based on previous assumptions presented in the methodology and this chapter.

By the indication from Li et al. (2018) previously presented, the nitrogen rate should be reduced when the additives are used. Equation 26 is used to calculate the nitrogen reduction potential in the case studies, which is the sum of the reduction achieved by the additive in each of the losses evaluated.

$$\boldsymbol{N_{reduction\,additive} = N_{additive-N2O} + N_{additive-VOL} + N_{additive-L}} \qquad (26)$$

$N_{reduction\ additive}$: total nitrogen reduction from additive use (kg N/ha)
$N_{additive\text{-}N2O}$: reduction of nitrous oxide emissions from additive use (kg N/ha)
$N_{additive\text{-}volatilization}$: reduction of nitrogen volatilization from additive use (kg N/ha)
$N_{reduction\ additive}$: reduction of nitrogen leaching from additive use (kg N/ha)

Equations 27 to 29 are based on the methodology presented in chapter 3.5.1 obtained from IPCC (2019a) and are used to calculate the respective reductions of each loss.

$$\boldsymbol{N_{additive-N2O} = N_2O\text{-}N \times AR_{N2O}} \qquad (27)$$

$$\boldsymbol{N_{additive-VOL} = (N_{VOL} \times AF) \times AR_{VOL}} \qquad (28)$$

$$\boldsymbol{N_{additive-L} = (N_L \times AF) \times AR_L} \qquad (29)$$

N_2O-N: amount of nitrogen released as nitrous oxide (kg N_2O-N/ha)
AR_{N2O}: reduction of nitrogen losses as nitrous oxide from additive use (%)
N_{VOL}: amount of nitrogen lost from the soil via volatilization (kg N/ha)
AF: adjustment factor for losses as leaching and volatilization in the Typical Farm
AR_{VOL}: reduction of nitrogen losses as volatilization from additive use (%)
N_L: amount of nitrogen lost from the soil via leaching (kg N/ha)
AR_L: reduction of nitrogen losses as leaching from additive use (%)

The data from the baseline scenario are used for the calculation of the equations. The reductions of the losses from additive use (AR_{N2O}, AR_{VOL}, AR_L) are obtained from scientific literature. If ranges are provided, the average value is used to depict the standard case. The upper value is used for the best case, as it depicts the highest reduction. The bottom value is used for the worst case because it indicates the lowest reduction. Literature reporting the changes in these three sources of losses is evaluated first. However, if no single source reporting on these three elements is identified, further applicable sources are incorporated into the equation. Thus, the estimation of the total nitrogen reduction can be the result of several different sources. The additives used in the Typical Farm are discussed in the focus group.

Fundamentally, it is assumed that the nitrogen reduction calculated does not affect yields. The added nitrogen from the reduced losses implies that the crop has the same supply of nitrogen as without the reduced rate. Nonetheless, this assumption is validated with a review.

3.6.3 Combination of strategies

The possibility to combine two or more mitigation strategies reveals the total GHG mitigation potential achievable in the Typical Farm for the crop selected and its associated costs. This supports the understanding of the total contribution toward climate change mitigation that can be expected from this sector. Moreover, assessing the combined effect of several strategies tackles a considerable short-coming in scientific literature, as this alternative is rarely analyzed (see chapter 2.4).

Evaluating the simultaneous implementation of the strategies is necessary as their assumptions or results may not cumulate. Synergies among elements of the strategies, potential incompatibilities or interdependencies may exist (MacLeod et al., 2015; Rotz et al., 2005; Samsonstuen et al., 2020; Teshager et al., 2017). For instance, a mitigation strategy may imply an additional farm operation that may compete for machinery with other strategies, but they can be combined in a single pass, lowering costs. Therefore, it is necessary to assess the joint implementation of the strategies to reveal these aspects (Schneider and McCarl, 2006; Kesicki and Strachan, 2011; Kesicki and Ekins, 2012).

Estimation of the reduction potential and cases

The evaluation of the combination of strategies begins with an assessment of the main changes of each individual strategy to identify limitations to their joint implementation. Farm operations, tillage and crop residue management are critically discussed at this stage as these are key in the definition of compatibility. The general possibility to combine the strategies into one or more scenarios is decided on the basis of this evaluation.

The interactions in the resulting combination of strategies assumed are analyzed next. Selected characteristics from the individual strategies are compared. This determines the possible differences that may arise from the joint implementation. To illustrate, two strategies affecting nitrogen rate may be only partially summable; thus, the resulting reduction may be smaller than the sum. The comparison includes not only alterations to the mitigation potential but also farm operations and input usage. The methodologies from the individual strategies, as explained in the corresponding chapters, are utilized to calculate the reduction in the combination. Only instances in which interactions may occur are discussed in this stage. Hence, the changes from individual strategies not presented in the combination are assumed to be unaffected.

3.7 Calculation of mitigation cost

The calculation of the mitigation costs resulting from the implementation of the mitigation strategy is obtained with Equation 30 (UNEP, 1998).

$$\boldsymbol{Mitigation\ cost} = \frac{\boldsymbol{Costs_{strategy} - Costs_{baseline}}}{\boldsymbol{GHG_{baseline} - GHG_{strategy}}} \times \boldsymbol{1{,}000} \qquad (30)$$

Mitigation cost:	cost to reduce one ton of GHG emissions (USD/t CO_{2eq})
$Costs_{strategy}$:	cost in the Typical Farm with mitigation strategy (USD/ha)
$Costs_{baseline}$:	cost in the Typical Farm in baseline scenario (USD/ha)
$GHG_{baseline}$:	GHG emission in baseline scenario (kg CO_{2eq}/ha)
$GHG_{strategy}$:	GHG emission with mitigation strategy (kg CO_{2eq}/ha)

The mitigation costs are presented at the end of each chapter evaluating the strategies. At the end of each case study, all the mitigation costs are compared as well as additional features.

Considerations regarding the adoption of the strategy

The discussion of the strategies in the Typical Farm also entails an assessment of context-specific alternatives to promote their adoption. Monitoring and policing of the strategies are also presented, as well as mechanisms to enforce their adoption. This discussion is conducted at the qualitative level, taking into consideration political and socioeconomic backgrounds. This discussion is presented with the mitigation cost.

Comparison of results from Typical Farm with literature

The mitigation potentials and costs calculated in this thesis are contrasted with results from literature to provide a reference to determine how they compare. As previously presented, the emissions from crop production and the mitigation strategies depend on crop, soil and climate factors. Thus, the mitigation potentials and costs from the literature reviewed and presented in the comparison are from studies reflecting conditions comparable to the respective Typical Farm. In this regard, the criteria are that the crop and the region must coincide. If no comparable results are identified, studies from similar regions or generated using a larger scale (e.g., national) are considered.

The studies and reports used in this comparison may have different assumptions, system boundaries and overall methodologies and do not necessarily match the specifications used in this thesis. Therefore, an exact comparison is not possible, yet it still provides a reference point. Nonetheless, the critical differences between the findings from literature and this study are presented.

Only studies that analyzed farm-level mitigation strategies are considered for the comparison. This includes individual strategies calculated in the context of bottom-up MACCs, since these upscale the results from farm-level calculations (see chapter 2.4). Furthermore, the comparison is based on the approach proposed to abate emissions (as explained in chapter 3.6.2) rather than on the specific implementation. For instance, the mitigation costs from oats as cover crops to sequester carbon on a Typical Farm can be compared with another study utilizing rye.

Nonetheless, literature in this regard is scarce and it is not possible to make a comparison for each case. Thus, only the strategies for which a comparison was possible are presented. The comparison is presented at the end of the assessment of each mitigation strategy estimated for the Typical Farm.

3.8 Definition of time horizons for policy advice

The mitigation strategies from each case study are ranked based on their mitigation cost. In the case of strategies that entail a transition and a stable stage (see chapter 3.5.6), a weighted average for a 100-year period is calculated (20-year transition period and 80-year stable stage). The transition stage of 20 years depicts the short-term perspective in which carbon sequestration occurs and the 100-year average, the long-term. This value is used solely to facilitate the comparison of the mitigation strategies since it is not within this project’s scope to make inferences regarding possible changes that may occur in this period. This 100-year period is chosen as it is usually used by IPCC in various projections - e.g., future trends in temperatures presented in Assessment Reports (IPCC, 2007, 2013) and their calculation of selected coefficients, including the GWP. The comparison of the results across the case studies shown in chapter 7.1 is also divided into short- and long-term perspectives utilizing the same approach.

3.9 Summary of approach in each case study

The same chapter structure and division are used in case studies. Each case study is introduced by presenting the general assumptions such as the exchange rate assumed for the cost calculation. The topic of each subchapter is:

Subchapter 1 presents the rationale to select the region within the country where the Typical Farm for the case study is located, following the criteria presented in chapter 3.1

Subchapter 2 briefly describes the climate in the region of the case study.

Subchapter 3 discusses agricultural production in the region and selected key features pertinent to the mitigation strategies. Aspects such as market structure and relevant policies and regulations governing arable farming are presented.

Subchapter 4 introduces the Typical Farm and evaluates its representativity in the regional context. The composition and realization of the focus group are explained (Chapter 3.4)

Subchapter 5 describes the assumptions used for cost and emissions calculations (Chapter 3.5). This is followed by the discussion of the individual mitigation strategies (Chapter 3.6) assessed by the focus group, and afterward, the presentation of the mitigation costs (Chapter 3.7) and considerations on the adoption, policing and enforcement of the strategies. The results of the Typical Farm are compared with findings from previous studies.

Subchapter 6 evaluates the possibility to combine the strategies and the limitations that may exist in that regard (Chapter 3.6.3). The costs and emissions of the combination of mitigation strategies are then evaluated.

Subchapter 7 ranks the mitigation strategies based on the costs.

Subchapter 8 presents the summary and the main findings of the case study.

Some approaches, concepts and assumptions are applicable in more than one case study. These are explained only in the first instance they are used. Thus, in all further occurrences, they are only mentioned and an indication to the chapter with the complete discussion is provided. In case there are differences in assumptions or other concepts, these are discussed again.

4 Case study 1: USA – Corn in Iowa

As presented in chapter 3.1, USA has been selected as one of the case studies based on its worldwide relevance as a corn producer and the presence of scientific partners cooperating with *agri benchmark* Cash Crop. This chapter introduces the assessment of each mitigation strategy for this Typical Farm, including the discussion of it in the focus group and the literature analysis used to validate the assumptions necessary for the calculation of the results.

4.1 Selection of the region

For the selection of the specific region within the USA where the case study takes place, the same criteria used for the selection of countries is applied. The fundamental idea is to choose a region where a significant share of the national production of corn is concentrated. The availability of members of the *agri benchmark* network (and respective farm data about the current production system) and their interest in cooperating in this project also were considered in the selection process.

Given that the country is divided into states and that most statistical data is presented at least at a state level, the preliminary selection of the study region for the Typical Farm is conducted based on state-level information. The USDA (2020) reports that the top three corn-producing states on average between 2017 and 2019 were Iowa with 17.9%, Illinois with 14.8%, and Nebraska with 12.1% of the total national production. Based solely on the principle of selecting a region that is relevant for the national production of corn, Iowa would be the selected state for the case study to be conducted.

With respect to having the support of local members of the network, *agri benchmark* has cooperated with experts from Iowa State University in the past. Furthermore, since 2008 they have been providing information for a Typical Farm located in northern Iowa. Given the relevance of Iowa as a corn-producing state and the scientific support that *agri benchmark* has in this state, the case study is conducted here.

4.2 Climate

Table 4.1 presents a summary of temperatures and precipitation for Iowa. Most of the rainfall is concentrated between May and July. Temperatures between December and February, on average, are below 0 °C and snow falls between the months of October and April. On average, the state receives 90 cm of snow between November and April, with 75% falling between the months of December and February. Frost can occur between late September and early May (Iowa State University, 2020). Due to the low temperatures in winter, it is common that the soil freezes from late November until February.

Table 4.1 Climatic data in Iowa (2010-2018)

Month	Temperature (°C)	Min. Temp. (°C)	Max. Temp.(°C)	Monthly rainfall (mm)	Number of Rain Days
Jan	-5.3	-9.8	-0.6	25.4	7.5
Feb	-2.8	-7.3	2.3	32.5	7.8
Mar	3.9	-1.3	9.4	58.4	9.7
Apr	10.8	5.1	16.8	98.0	11.2
May	16.7	11.2	22.4	120.4	12.5
Jun	22.1	16.7	27.6	125.5	11.5
Jul	24.6	19.3	29.8	113.5	10.1
Aug	23.4	18.2	28.8	104.9	9.1
Sep	18.6	12.9	24.5	77.5	8.4
Oct	11.5	6.1	17.3	67.1	8.9
Nov	3.9	-0.8	8.8	55.6	8.3
Dec	-3.4	-7.8	1.1	36.1	8.4

Source:NOAA (2020)

4.3 Background and key features of arable farming in Iowa

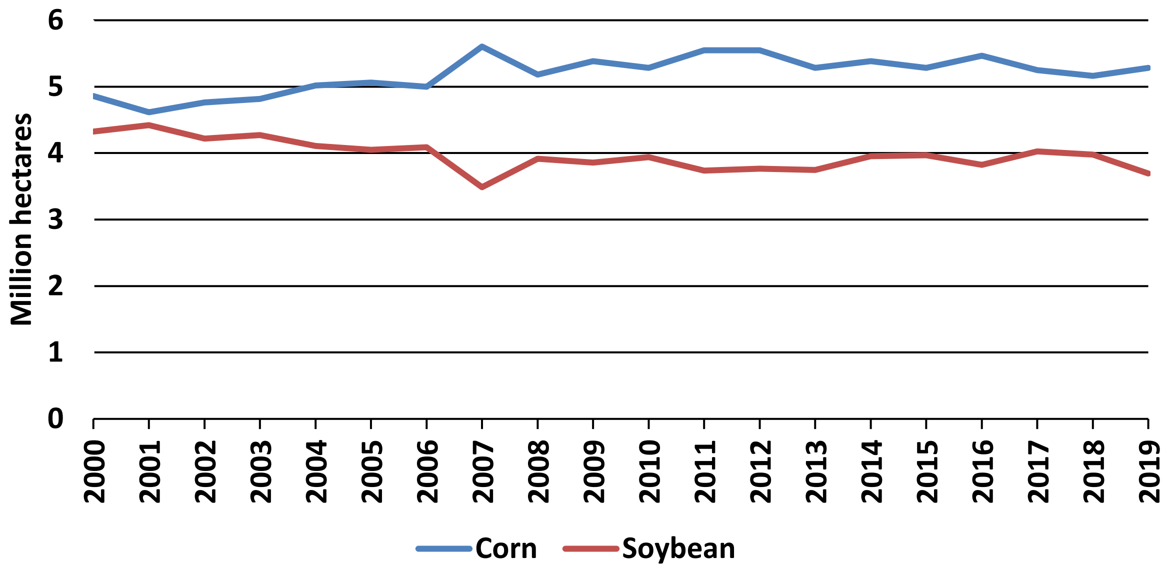
Iowa is the largest exporter of grain corn from all states (USDA, 2021a). The state, on average, had 9.8 million ha dedicated to cropland from 2017 to 2019, approximately 70% of the state's total land. According to the USDA (2020), corn acreage in 2019 was approximately 5.5 million ha. The crop is grown on 56% of the state's cropland followed by soybeans with 3.8 million ha or 39% of the cropland. About 2% of the land is dedicated to the production of hay and corn silage and the rest is divided among crops with small shares such as oats and sweet corn.

The high share of corn and soybeans in the state's cropland represent the most common crop rotation practiced in Iowa: corn followed by soybeans, each as an annual crop. The most usual alternatives to this crop rotation are corn-corn-soybeans and continuous corn rotations, options which have become more popular in the state over the past years because of relatively high corn prices as a consequence of increased demand for corn as feed and for ethanol (Al-Kaisi et al., 2015).

The national government in 2005 introduced a law making the blend of ethanol into vehicle fuel mandatory. This led to the construction of ethanol mills, which created a new source of demand for corn. Since its introduction, the national demand for corn for ethanol quadrupled between 2005 and 2019 (USDA, 2021b). It is estimated that this sector currently consumes approximately 50% of the state's total production of corn (U.S. Energy Information Administration, 2020) In this regard, Iowa is the nation's biggest producer of ethanol, generating approximately 26% of the country's supply in 2019.

On average, corn's acreage increased between 2000 and 2019 by approximately 13%, while soybeans' acreage decreased by 12% during the same period (USDA, 2020). However, this trend has not been constant across all years as in some cases the acreage of soybeans increased. The changes in acreage are presented in Figure 4.1.

Figure 4.1 Acreage of corn and soybeans in Iowa from 2000 to 2019



Source: own elaboration based on USDA (2020)

Another relevant driver of this change in acreage has been the yield developments of these crops. Plant breeding, better management and production practices have increased corn's yield proportionally greater compared with soybeans over the past years (Fischer and Edmeades, 2010). In the early 2000s, the yield ratio for soybeans-corn was 1:3, that is, for each ton of soybeans grown, three tons of corn could be produced instead. Based on the averages from 2018 and 2019, the yield ratio has increased to 1:3.6 (USDA, 2020).

Hofstrand (2020a, 2020b) provides an estimation on the average production costs of soybeans and corn in Iowa. He estimates that between 2000 and 2019, these costs increased by 58% for soybeans, but only 46% for corn production.

For the same period, and taking the year 2000 as baseline, the average annual increase in the price was 6.6% for corn and 6.2% for soybeans (USDA, 2020). This means that the price of corn grew comparatively more than the price of soybeans. Consequently, the less-than-proportional increase in the production costs of corn compared with soybeans, paired with an over-proportional increase in the price and yield of corn, made the production of corn more profitable, which has further promoted its production in the region.

4.4 Typical Farm and focus group

The preexisting Typical Farm is used as a baseline for the discussion in the focus group. Adjustments to the baseline scenario as well as changes required for the mitigation strategies follow the approach explained in chapter 3.4.1.

4.4.1 Typical Farm and its representativity

To assess the representativeness of the Typical Farm and the results generated in the focus group discussion, publicly available survey data and reports are presented here. As explained in chapter 3.4, Typical Farms are not representative in the statistical sense. This means they do not necessarily represent the statistically average farm, but rather depict a farm that produces the bulk of the products. Consequently, differences between statistical data and *agri benchmark* data can be expected.

Nevertheless, an initial comparison as a reference could be made based on the size of the farm. According to the USDA's Agricultural Census (2020), the average farm size in Iowa is 145 ha, compared with the 728 ha of the Typical Farm. However, USDA's value includes all types of farms, not just full-time managed, arable farms, as is the case with the Typical Farm. Since livestock and horticultural farms usually are smaller in land size than arable farms, their inclusion in this statistic distorts the interpretation of this value. Nonetheless, the USDA's Agricultural Census reports the acreage harvested and divides it into different farm sizes. In this statistic, the Typical Farm belongs in the category of farms that operate between 614 and 1,018 ha (1000 to 1,999 acres). This farm size category operates the largest share of all farm size classes, managing approximately 25% of Iowa's total arable acreage. The combined size classes of smaller farms manage in total 53% of the total arable land, while larger farms operate 21%. No information is provided on the average yields for each segment. In this regard, assuming there are no significant differences within or between the segments, it can be inferred that the Typical Farm belongs to a size category that produces a considerable share of Iowa's total crop production.

The Typical Farm is operated under a reduced tillage system. Statistical data indicates that 43.3% of the acreage in Iowa is under this tillage regime, 35.1% is operated as no-till, and only 21.5% as conventional (USDA, 2020). Hence, in terms of tillage management, the Typical Farm belongs to the most common category.

The crop rotation is corn–soybeans, which is common in this state based on the Agricultural Census (USDA, 2020). Iowa State University (2016, 2017, 2018) provides estimations of yields based on data from the USDA's Agricultural Census and of agricultural inputs based on their calculations. These estimations are based on data from the entire state. For this comparison, the data is averaged for the 2016-2018 period. The average is used to depict a "normal" year in the focus group discussion.

Table 4.2 Comparison of Typical Farm with estimations from Iowa State University

	Corn			**Soybean**		
	Typical Farm	**Estimation**	**Difference***	**Typical Farm**	**Estimation**	**Difference***
Yield (t/ha)	12.7	12.6	0.8%	3.8	3.7	3.0%
Seed (USD/ha)	245.4	299.8	-18.1%	127.7	130.4	-2.0%
N (kg/ha)	180.8	146.8	23.1%	0.0	0.0	0.0%
P_2O_5 (kg/ha)	79.6	84.1	-5.3%	48.3	49.3	-2.1%
K_2O (kg/ha)	63.4	67.3	-5.7%	91.7	93.0	-1.5%
Herbicide (USD/ha)	75.7	81.1	-6.6%	90.6	89.4	1.3%

*Considers the estimation as the reference point to compare it with the Typical Farm.
Source: own estimation based on data from *agri benchmark* agri benchmark Cash Crop (2022) and Iowa State University (2016, 2017, 2018).

Table 4.2 shows that the yields obtained by the Typical Farm are close to the yields reported by the Census (USDA, 2020). The Typical Farm spends approximately 18% less on seeds for corn compared with the estimated costs. Since the census does not provide detailed information on traits or other characteristics other than whether or not it is transgenic, it is not possible to explain this difference.

Nitrogen use in corn production by the Typical Farm is 23% higher than in the estimation. However, this estimation is partly based on the values reported by the USDA's Agricultural Survey, which considers only commercial fertilizer, including purchased manure[10]. Hence, manure produced in the grower's operation is not considered in this statistic, indicating that a share of the nitrogen applied is missing in the estimation. In the case of Iowa, it was estimated that livestock production in 2012 generated enough manure to cover 30% of the state's total nitrogen requirement for corn production (Andersen and Pepple, 2017). This manure must be applied to the field, or the grower risks having to pay fines. Since phosphorus and potassium also are present in the manure, the same problem is valid for these nutrients. Therefore, it can be inferred that corn production in Iowa received a significant dosage of fertilizer that is not reflected in USDA's Agricultural Survey, which was partly the base used for the estimation. Therefore, a comparison of the use of fertilizers between estimation and the Typical Farm which, by definition, includes all fertilizers used and not just purchased ones, is not adequate.

The difference in the cost of herbicides is small, with the Typical Farm spending a little less than the estimation. No details on the dosage or active ingredients are provided; hence, no further assessment is possible. Neither the Typical Farm nor the estimations report the use of any other crop care product besides herbicides; therefore they are not presented.

The differences between the estimation from Iowa State University and the Typical Farm can be considered critical, especially because their estimation of nitrogen rate is partly based on data that does not necessarily consider all the nitrogen applied. This discrepancy in fertilization limits the assessment of representativity of the mitigation strategies, as fertilizer plays a central role in GHG mitigation strategies. Nonetheless, in the absence of more detailed data, it is not possible to derive further inferences and the representativity of the results is provided by the members of the focus groups that operate in the region selected.

4.4.2 The Iowa focus group

The focus group took place in July of 2019, and it was composed of six growers, an agricultural consultant, the scientific partner responsible for the Typical Farm in Iowa and the researcher conducting this project. A literature review was conducted in preparation of the discussion. The first step was the presentation of the project to the participants to explain the objective of the research project and the goal of the focus group discussion.

The second step consisted of discussing the Typical Farm. For this purpose, the preexisting Typical Farm in Iowa from *agri benchmark*'s dataset was used as a base. An average of the data from 2016 to 2018 was used to describe a "normal" year. Key elements regarding inputs, yields and agricultural operations were presented and discussed with the participants. The outcome of this discussion was that the data of the Typical Farm already depicts a common farm in this region, confirming the conclusion from chapter 4.4.1. This data was considered the status quo for the region and the starting situation from which the mitigation strategies were discussed.

The last step of the focus group consisted of discussing the mitigation strategies for corn production. For each strategy, the changes in inputs, yields and farm operations necessary for the implementation of the mitigation strategy were discussed in depth. The possible effects on the rest of the crop rotation were evaluated as well. Additional factors such as availability of inputs or technical knowledge were included in the discussion. Each strategy was discussed until an agreement was reached among all the participants. Targeted interviews and visits were conducted to further discuss aspects that help understand the context and implications of the strategies. A literature review was conducted to compare results and validate the data.

[10] It was not possible to find statistics on the value or volume of manure traded in the state. Consequently, it is not possible to infer the share of manure traded at state-level. In this regard, the focus group commented that they are not aware of any supplier of commercial manure.

4.5 Mitigation strategies analyzed

In this chapter, the implications at the farm level of each GHG reduction strategy for corn production on the Typical Farm in Iowa are presented and discussed. The GHG mitigation strategies are assumed to be implemented independently from each other. Hence, each strategy is assessed individually in its own chapter. This individual assessment is compared against the baseline scenario and assumes that none of the GHG mitigation strategies discussed is being used. Therefore, the implications and results of each strategy are applicable only for the chapter in which it is discussed and are not valid for subsequent mitigation strategies. The possibility to combine strategies and their implications are presented in chapter 4.6. A comparison of the results from all strategies assessed in this case study is presented in chapter 4.7.

For each strategy, the changes in the costs derived from the adjustments needed to implement the GHG mitigation strategy are calculated and explained. The corresponding GHG reduction potential per hectare and per ton of product are estimated as well. The results per ton of product are presented only if the mitigation strategy has an effect on the yields. Otherwise, these results are presented only in Appendix B, as they would not portray additional information compared with the results per hectare. The reduction in the emissions per hectare, along with the change in costs, are used to calculate the GHG mitigation cost per ton of CO_2.

The strategies are assigned a code composed of two elements: initials to denote the country and a number. The baseline scenario is assigned the number zero, hence, this scenario for the Typical Farm in Iowa is coded as US-0. The codes for each strategy are shown in the title of the chapter for the mitigation strategy.

The detailed costs and GHG emissions of the Typical Farm are presented in chapter 4.5.1, which shows the baseline scenario (US-0). In the tables presenting the results of the strategy assessed, only the categories of costs and emissions that are affected by the strategy are kept disaggregated. All the categories that are not affected by the mitigation strategies are grouped in the category named “remaining.” Consequently, this category entails the elements that are the same as in the baseline scenario.

Each strategy is calculated using a standard, best-case and worst-case calculation. These cases vary on the assumption of the GHG mitigation potential attainable with the implementation of the strategy. The three cases are presented in the tables showing the results in costs and mitigation potentials. This also implies that the mitigation costs are estimated using the three cases. Appendix A contains the tables presenting the assumptions used for the calculation of the mitigation costs in this case study[11].

4.5.1 Status quo – baseline scenario (US-0)

The Typical Farm in Iowa, as confirmed by the focus group, depicts the prevailing status quo in terms of input use, input costs, farm operations and yields for corn production for the region. This data is used to determine the GHG emissions from the current system. The mitigation strategies are compared against the status quo to determine the reduction potential of GHG emissions and the associated costs.

The costs of corn production in the baseline scenario are shown in Table 4.3. Some mitigation strategies implemented in corn affect soybean production; hence, the costs of this crop are shown as well. These are the costs against which the changes resulting from the implementation of the mitigation strategies are compared.

---

[11] Input costs are presented in Table A.1, coefficients for the GHG calculation in Table A.2, inputs used in corn in Table A.7, inputs used in soybeans in Table A.8, description of machinery in Table A.11, and farm operations in Table A.13.

Table 4.3 Costs of corn and soybean production for Typical Farm in Iowa (USD/ha)

	Corn	Soybean
Seed	245	128
Fertilizer	245	100
Pesticides	76	69
Liming	0	0
Diesel	48	22
Labor	37	20
Machinery	357	267
Contractor	0	0
Total	**1,008**	**606**

Source: own estimation.

In total, the production costs are calculated as 1,008 USD/ha for corn and 606 USD/ha for soybeans. The machinery cost is obtained from the preexisting Typical Farm already included in *agri benchmark*'s database, which is calculated as indicated by Nehring (2011). Thus, this value is not obtained from the focus group from this case study but is still provided as an indication of the average farm-level machinery costs in Iowa. Nonetheless, the changes in machinery costs resulting from the implementation of the GHG mitigation strategies are calculated as described in chapter 3.4.1. Under the current system, the Typical Farm does not use agricultural contractors for any operations. Therefore, it has no costs in this category. Similarly, the Typical Farm does not apply lime in this scenario.

Emissions per hectare in the baseline scenario

As explained in 3.5.1.2, the indirect emissions are modified using an adjustment factor. This factor is applied to change the nitrogen losses estimated by IPCC (2019a) so the total nitrogen removals and losses match the nitrogen available in the system. Along with the application of fertilization, nitrogen fixation by legumes is considered as a nitrogen input available in the system. In the case of soybeans, based on the meta-analysis by Ciampitti and Salvagiotti (2018), it is estimated that soybeans fixate 59 kg N/t harvested[12]. The yield reported by the Typical Farm is 3.8 t/ha; hence, the crop is assumed to fixate a total of 224 kg N/ha. Since no further nitrogen is applied, this value represents the total nitrogen available. To determine the nitrogen removal of soybeans, the same yield is used but the moisture content, which is assumed to be 13%, is deducted to obtain the crop's dry matter. It is assumed that the dry matter of soybeans has a nitrogen content of 6.6% (Ciampitti and Salvagiotti, 2018). This results in a removal of 218 kg N/ha.

In corn, the supply of nitrogen is provided only by the anhydrous ammonia, which totals 181 kg N/ha. Based on Córdova et al. (2019), it is assumed that soybeans' nitrogen fixation does not leave a nitrogen credit that corn can take up. Sawyer and Mallarino (2008) report that the average nitrogen content in the grain is 1.31% of the dry matter. The moisture of the grain is 15.5%, based on the focus group. Considering a yield of 12.7 t/ha and subtracting from that value the moisture content, 140 kg N/ha are removed from the system via harvest. The moisture and nitrogen content assumed are intrinsic to the crop and are not affected by the mitigation strategies. Consequently, the values used in the baseline scenario (US-0) are valid for all the strategies analyzed in the Typical Farm in Iowa.

For the estimation of the emissions from the categories direct and indirect with IPCC's (2019a) approach, the climate zone from the region of the Typical Farm is needed. Based on Sayre et al. (2020), the climate is

[12] Ciampitti and Salvagiotti (2018) determine soybean yield (13% moisture) as a function of the nitrogen fixation rate. Each kilogram of nitrogen fixated generates a yield of 17 kg of soybean. This is equivalent to 59 kg of N fixated in the soil per ton of soybean produced.

assumed as Cool Temperate with a Moist regime. Table 4.4 presents the emissions of the Typical Farm in the baseline scenario.

Table 4.4 Emissions from corn and soybeans for Typical Farm in Iowa in baseline scenario

	Corn		Soybean	
	Hectare ($kg\ CO_{2eq}/ha$)	Product ($kg\ CO_{2eq}/t$)	Hectare ($kg\ CO_{2eq}/ha$)	Product ($kg\ CO_{2eq}/t$)
Direct	1,647	130	133	35
Indirect	200	16	35	9
Liming	0	0	0	0
Urea	0	0	0	0
Manufacture	606	48	59	16
Diesel	245	19	111	29
Land use	0	0	0	0
Total	**2,698**	**213**	**338**	**89**

Source: own estimation.

A summary of the elements entailed in each emission category is presented in chapter 3.5.7. The total GHG emissions of corn in the baseline scenario are 2,698 kg CO_{2eq}/ha. For soybeans, the emissions are 338 kg CO_{2eq}/ha. Given that the farm does not use lime or urea in either crop, the emissions from these categories are zero. The Typical Farm has been operated with the same tillage regime and input of biomass for longer than 20 years. Consequently, following IPCC (2019a) methodology, it is assumed that there is no net change in the carbon content of the soil (see chapter 3.5.6). Thus, no emissions from land use are calculated.

4.5.2 Optimization of nitrogen rate (US-1)

Optimizing the nitrogen application rate can reduce emissions from the soil by decreasing the fertilizer input, reducing the emissions from its production. The potential reduction in the rate can be calculated using the NUE and comparing it to the current fertilization rate. To calculate the NUE, it is necessary to determine the nitrogen removal via harvest. As shown in 4.5.1, for corn, it is estimated that 140 kg N/ha are removed. Currently, the fertilization rate of nitrogen is 181 kg/ha. This implies that the farm operates at a NUE of 78%. No specific NUE values could be found for this region after a literature review. Hence, considering Brentrup and Palliere (2010) standard values, a NUE of 85% is defined as ideal (see chapter 3.6.2.1). In this case, the grower should be able to reduce the nitrogen input to 165 kg N/ha, meaning that, in theory, a possible reduction potential of approximately 15 kg N/ha can be achieved without negatively affecting corn yields.

Following the approach by Brentrup and Palliere (2010), the best-case scenario assumes an NUE of 90%. This value indicates the highest nitrogen reduction potential achievable without negatively affecting the crops and, consequently, the highest GHG reduction potential. The worst-case scenario assumes an NUE of 80% and signifies the lowest nitrogen and GHG reduction potential. The nitrogen reduction potential of these two cases as well as the standard case are presented in Table 4.5. These values also are used to estimate the adjustment factor applied to indirect emissions (see chapter 3.5.1.2). The difference between the fertilizer rate calculated for each case and the rate in the baseline scenario reveals the fertilizer reduction.

Table 4.5 Reduction of nitrogen rates based on optimal NUE for Typical Farm in Iowa

Case	N removed via harvest (kg of N/ha)	Fertilizer rate (kg of N/ha)	NUE (%)	N reduction potential (kg of N/ha)
Baseline	140	181	78	-
Standard	140	165	85	15
Best	140	156	90	25
Worst	140	176	80	5

Source: own estimation based on Brentrup and Palliere (2010).

To estimate the NUE it is necessary to consider all the nitrogen sources. In this regard, corn is grown after soybeans, a legume, and the nitrogen fixed by the soybeans should be accounted toward the supply of nitrogen for corn. However, soybeans' fixation rate is assumed to be only enough to cover its own requirements (Córdova et al., 2019). Consequently, it does not leave a nitrogen credit in the soil that corn can utilize. Hence, the NUE estimation is conducted considering only the synthetic nitrogen applied to corn.

In the fertilization program of the baseline scenario, the nitrogen is applied as anhydrous ammonia in spring, usually one week before seeding. To study a reduction of nitrogen applied, the focus group agreed that splitting the applications into two could be a feasible approach, each pass with the same rate of fertilizer. They expressed that it could be possible to include the additional application approximately one month after seeding. Scharf et al. (2002) and Roy et al. (2014) show that splitting nitrogen fertilization to include a second application later in the season does not negatively affect corn yields. They estimate that the second application can occur until the plant has eight leaves, which occurs on average 40 days after seeding, without altering yields. Hence, the application can take place at any point within this interval. Consequently, as a strategy to reduce the total nitrogen rate used in corn, a second pass to apply anhydrous ammonia is assumed.

This second application of fertilizer is as side-dress, that is, the anhydrous ammonia is applied into the soil close to the plant, between the rows. The focus group mentioned that there is a risk of damaging the crop, as anhydrous ammonia potentially can injure the roots and stem by producing chemical burns. However, with proper adjustment of the machinery to ensure a safe distance from the roots, the damage can be avoided, and the yields are not negatively affected (Woli et al., 2014).

The additional pass can be conducted using the same equipment utilized in the first application. Hence, no new machinery is required to split the applications of nitrogen. It is assumed that the additional application does not cause further depreciation at the farm level, since the machinery is still operated within the planned yearly threshold. Hence, only a redistribution of the depreciation cost occurs and the total for the grower does not change (see chapter 3.4.3). However, the additional pass implies added costs in repairs and maintenance as well as higher use of diesel and labor.

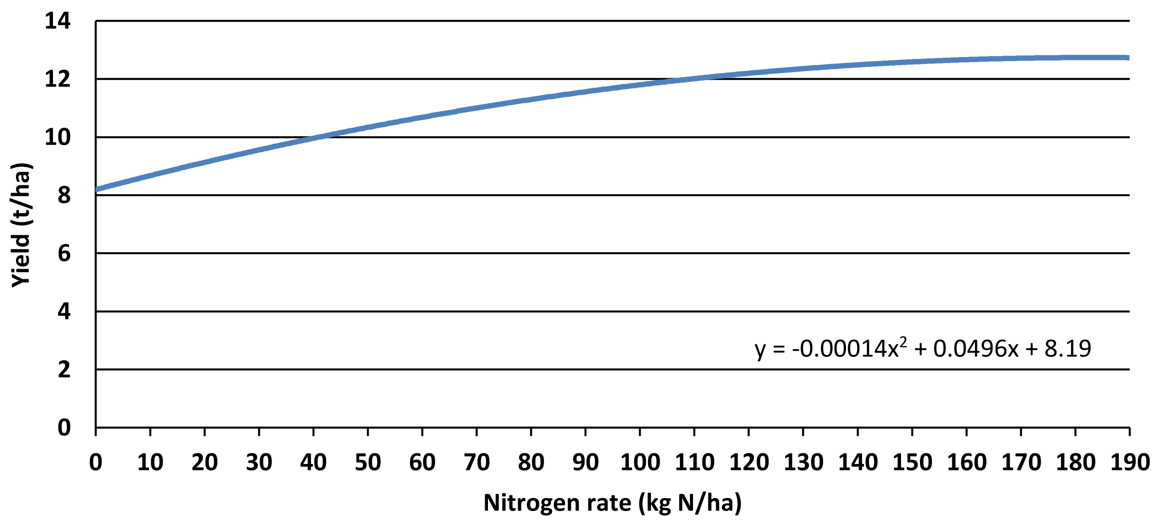
Nevertheless, the additional operation could compete with the operations needed in soybean production. Under the current schedule of operations, the application of fertilizer could be performed without directly affecting other operations, as expressed by the focus group. However, they also comment that when planning the season, they include a time buffer by planning more time for the operations than what is needed. They do this in case operations have to be delayed for unforeseen reasons, such as adverse weather or unexpected repairs. The additional application of nitrogen would considerably reduce the time buffer that growers have, as explained by the focus group.

The next operation in the schedule of the Typical Farm is the application of the herbicide glyphosate to soybeans. Thus, in the case of having to delay the second application of anhydrous ammonia, the spreading of this herbicide may have to be delayed as well. In this case, the grower has the option to increase the concentration of the herbicide application. This would increase the share of the weeds that would be killed by the herbicide (Westra et al., 2008). It is likely that the increase in the concentration of the herbicide would be comparatively small relative to the normal rate used. Hence, the increase in the costs of the herbicide would be minor as well. To exemplify, increasing the dosage of the herbicide by 20% would imply only approximately 10 USD/ha more, based on the standard glyphosate application used in the Typical Farm. Alternatively, the rate of the herbicide could be left unchanged in the delayed application. In this case, the grower could perceive a yield loss compared with the case in which the application of the herbicide is conducted without a delay. This yield reduction would be caused by the additional competition the crop would have. The yield loss would result from the time between the date of the planned herbicide application (without delay) and the actual date of the application (with delay). Consequently, it can be expected that the yield loss would be minimal and the corresponding loss of revenue would be small as well. Accordingly, considering that both alternatives to manage a delayed application have a small negative economic impact, added to the fact delaying the application theoretically would not happen frequently, the economic impact of the competition for machinery is considered negligible. Consequently, for the estimation of the costs of this mitigation strategy, these costs are assumed to be insignificant.

Outsourcing the second application of nitrogen to a contractor theoretically would nullify the opportunity cost of the grower's machinery caused by the additional pass and increase the growers' time flexibility but also increase cash costs. However, the focus group stated that there are contractors offering this service in the region but, based on their own experience, the availability is scarce. Consequently, this option is not considered.

The participants of the focus group expressed concern regarding the reduction as they would be worried that the rate may not be enough to achieve the expected yield. To validate the effect that the nitrogen reduction could have on corn yields, Sawyer et al. (2006) are used as a reference. They provide a nitrogen response curve[13] that presents the effect marginal variations in nitrogen application have on corn output. The results are based on field trials in 226 sites distributed across Iowa and are calculated as a percentage of the potential yield.

Figure 4.2 Estimated corn yield under different nitrogen fertilization rates in Iowa



Source: own estimation based on Sawyer et al. (2006).

[13] After several attempts at contacting the authors, it was not possible to obtain the raw data used for the regression. Therefore, the data points were extracted from the curve presented by the authors in the report. The points were used to obtain the regression presented. The curve generated was compared graphically with the original to assess the result.

Figure 4.2 presents the theoretical corn yields that the Typical Farm would obtain based on Sawyer et al. (2006) nitrogen response curve. The nitrogen rates in the baseline scenario and the calculated reduced rates are used in the formula to determine the marginal change in yields. By reducing nitrogen application from 181 kg/ha to 165 kg/ha to achieve a NUE of 85%, corn yield would be reduced by 0.04 t/ha. An NUE of 80% generates a yield loss of 0.01 t/ha. For the case of 90% NUE, the reduction is estimated to be 0.1 t/ha, or 0.8% less yield. These yield reductions are considered negligible and, therefore, confirm the assumption that adapting the nitrogen fertilization rate utilizing the NUE does not negatively affect yields for this Typical Farm. Consequently, no yield loss is assumed for this mitigation strategy.

Total costs from implementing the mitigation strategy

Table 4.6 presents the changes in the costs derived from reducing the nitrogen use in corn production on the Typical Farm. These changes reflect the reduction in fertilizer costs due to optimizing the fertilization rate, as well as the additional machinery, diesel and labor costs from splitting the applications in two.

The reduction in fertilizer rates reduces costs, which differ across the cases based on the NUE assumed. The reduction ranges from 19 USD/ha in the best case to 4 USD/ha in the worst case. In the standard case, the reduction is 12 USD/ha. In the standard and best cases, this reduction is enough to compensate for the additional costs the grower incurs conducting an additional application of fertilizer. This additional pass is estimated to cost 8 USD/ha, which includes labor, diesel and repairs and maintenance and implies the same costs across the cases. Consequently, in the standard and best cases, which assume 85% and 90% NUE, respectively, the GHG mitigation strategy would imply a decrease in the costs of producing corn on the Typical Farm. Only in the worst case, which assumes an NUE of 80%, would the decrease in nitrogen rate not be enough to cover the costs of the additional pass to spread fertilizer. Hence, in the worst case, the mitigation strategy would increase the cost.

The total cost of the mitigation strategy implies a reduction of 4 USD/ha in the standard case and a reduction of 11 USD/ha in the best case. In the worst case, the total increase in costs is 5 USD/ha. These are the values used to determine the mitigation costs.

Table 4.6 Corn costs with optimized nitrogen fertilization rate for Typical Farm in Iowa (USD/ha)

	Baseline scenario	Standard case		Best case		Worst case	
		Strategy	Change	Strategy	Change	Strategy	Change
Fertilizer	245	233	-12	226	-19	241	-4
Diesel	48	51	3	51	3	51	3
Labor	37	39	2	39	2	39	2
Machinery	357	360	3	360	3	360	3
Remaining	321	321	0	321	0	321	0
Total	**1,008**	**1,005**	**-4**	**997**	**-11**	**1,013**	**5**

Source: own estimation.

Emissions per hectare from implementing the mitigation strategy

Table 4.7 shows the differences in the emissions of the baseline scenario compared with the mitigation strategy. The reduction in the emissions from this strategy comes from the reduced use of nitrogen fertilizer, which lowers the N_2O emissions from the soil depicted in the category "direct." This reduction ranges from 184 to 37 kg CO_{2eq}/ha in the best and worst cases, respectively. In the standard case, the reduction in emissions is 115 kg CO_{2eq}/ha. The emissions caused by leaching and volatilization represented in the category “indirect” are reduced as well. The reduction is 82 kg CO_{2eq}/ha in the standard case, 130 kg CO_{2eq}/ha in the best case and 27 kg CO_{2eq}/ha in the worst case.

The emissions allocated to the manufacture of fertilizers are reduced as well. This reduction ranges from 74 to 15 kg CO_{2eq}/ha in the best and worst cases. In the standard case, the reduction is 47 kg CO_{2eq}/ha. The magnitude of the change in direct, indirect and manufacture emissions is different in all cases as it depends on how much nitrogen is reduced by the NUE approach.

Liming, urea and land use are grouped in the category “remaining.” None of these categories is affected by the implementation of the GHG mitigation strategy. Hence, as in the baseline scenario, no GHG emissions are released from these sources.

The additional pass to spread the fertilizer releases additional emissions from diesel combustion (18 kg CO_{2eq}/ha) but these are comparatively low. Considering an NUE of 85% in the standard case, this strategy reduces GHG emissions by 226 kg CO_{2eq}/ha. For the best and worst case scenarios, the GHG reduction ranges from 371 to 62 kg CO_{2eq}/ha, respectively.

Table 4.7 GHG emissions from corn with optimized nitrogen fertilization rate strategy for Typical Farm in Iowa (kg CO_{2eq}/ha)

	Baseline scenario	Standard case		Best case		Worst case	
		Strategy	Change	Strategy	Change	Strategy	Change
Direct	1,647	1,532	-115	1,463	-184	1,610	-37
Indirect	200	118	-82	69	-130	173	-27
Manufacture	606	560	-47	532	-74	591	-15
Diesel	245	262	18	262	18	262	18
Total	**2,663**	**2,472**	**-226**	**2,327**	**-371**	**2,636**	**-62**

Source: own estimation.

Mitigation costs and considerations on the adoption, monitoring and enforcement of the strategy

In total, optimizing nitrogen fertilization to lower GHG emissions would generate a cost of -16 USD/t CO_{2eq} in the standard case, -29 USD/t CO_{2eq} in the best case, and 74 USD/t CO_{2eq} in the worst case. In the standard and best cases, the mitigation strategy would be a win–win scenario, which is implied in the negative cost. This signifies that the strategy not only reduces GHG emissions but also lowers the cost of producing the crop. The implications of the win-win scenario are discussed in chapter 8.

Since this strategy could be adopted by growers without the need to make investments, its implementation on a large scale and in a short time frame should be feasible. Given the relevance of nitrogen fertilization to obtain high corn yields (see chapter 3.6.2.1), growers may be reluctant to adopt this strategy, as they may perceive it as an additional risk. An approach that could be used to address this concern would be to promote a stepwise adoption, in which two aspects could be adopted: acreage and nitrogen reduction. A stepwise adoption based on acreage would mean that growers test the reduction of nitrogen only on a share of their land. Once the approach has been tested, it could be expanded gradually each year to cover a larger share. A stepwise adoption of the nitrogen reduction would mean that the grower reduces nitrogen fertilization rate by a share of what this mitigation strategy indicates. This share could be increased gradually until the recommended nitrogen reduction rate is achieved. Furthermore, a combination of these two approaches could be promoted as well.

An additional measure that could be used to reduce the perceived risk by the grower is to provide an insurance. This service could cover the cases in which splitting of the nitrogen application to reduce the rate does negatively affect yields - or instance, in years in which weather conditions delay the second application of nitrogen or any subsequent operation, such as the use of pesticides in soybeans, as previously explained. To obtain a reference for the estimation of the yield loss caused by this GHG mitigation strategy, a comparison with field trails could be conducted. These field trials would maintain the baseline scenario; that is, no splitting of nitrogen application and no reduction of nitrogen rate. By comparing the difference between the grower's yields and the field trials, the compensation could be calculated.

Monitoring the use of this strategy likely would require field visits. Aerial images obtained from drones flown on the fields can be run through an algorithm that helps determine the soil's nitrogen content. This process should be supported by soil sampling to calibrate the model for different soil types (Hively et al., 2009). According to Wójtowicz et al. (2016), the sensing technology from satellites is not yet able to deliver reliable results on this type of data, although it may become a feasible approach in the future. Consequently, it can be assumed that monitoring the implementation of this strategy likely would entail high costs as yearly and on-site testing would be required for proper measurement.

A limitation of using remote sensing is that the cameras need to capture enough data on the soil. Hence, this technology is applicable only as long as the crop's canopy does not block the view of the soil. Measuring the nitrogen content of the crop is not a feasible indicator of the soil's nitrogen amount. Assuming the nitrogen supply has not limited the crop's development, it may not be possible to make accurate inferences on the amount of nitrogen remaining in the soil. Hence, it is not possible to determine how much nitrogen has been applied with current technology.

The implementation of a tax on nitrogen would promote an increase in the efficiency of fertilizer usage. Increasing the fertilizer's price would raise the fertilizer–crop ratio, which implies that growers would need to use nitrogen more efficiently by lowering the nitrogen rate. Hence, growers likely would adjust the timing of the application or adopt new technologies; for instance, equipment with variable-rate application (Zhang et al., 2015b).

Another approach that could encourage growers to increase the NUE as estimated in this strategy is to limit the amount of fertilizer they can purchase. However, considering the size of the region and country, the variation of yields and their development, variation among plots and possible crop rotations, among other aspects, using this approach to enforce the strategy would entail high costs for the controlling entity. Consequently, it is not considered as an efficient alternative.

Comparison of results from Typical Farm with literature

McNunn et al. (2020) simulated the GHG emissions from improving the timing of nitrogen fertilization in corn production across 11 states in the Corn Belt, including Iowa. In total, they estimate a reduction in the range of 97 to 510 kg CO_{2eq}/ha. The simulations from the authors include different baselines and alternatives to implement this mitigation strategy, including the replacement of fall fertilizer applications. This practice is not used on the Typical Farm in Iowa. Nitrogen applied in the fall has a higher risk of being lost as leaching and emissions compared with spring applications. Consequently, fall fertilizer rates have to be higher to compensate for increased nitrogen losses (Tenuta et al., 2016). Thus, the baseline assumed in the study by McNunn et al. (2020) has greater emissions than the baseline in the Typical Farm, which spreads the fertilizer only in spring. Therefore, by improving the timing of the application, the emission reduction potential theoretically is higher than in the Typical Farm.

Millar et al. (2010) calculated that a reduction of 130 to 490 kg CO_{2eq}/ha could be achieved by reducing nitrogen oversupply in corn in Iowa. Similarly, the report by ICF (2013) estimated that a reduction of 74 to 400 kg CO_{2eq}/ha is possible in the Corn Belt. Both studies modeled the fertilizer reduction based on the yield expected using state-level data for Iowa, including historic yield data and fertilizer applications. None of these studies includes emissions from the manufacture of fertilizer, which would have increased the mitigation potential calculated in the studies, as each unit of fertilizer reduced would imply a higher reduction. However, diesel usage from additional applications is not accounted for, which would increase the emissions from the strategy.

In comparison, the mitigation potential in the Typical Farm in Iowa ranges from 62 to 371 kg CO_{2eq}/ha in the worst and best cases. It could be argued that the worst case of the Typical Farm is similar to the bottom of the range from ICF (2013) and McNunn et al. (2020). The best case in the Typical Farm is comparable only to the upper value from the estimation by ICF (2013). The remaining two studies indicate a larger mitigation potential than that of the Typical Farm.

ICF (2013) calculated the mitigation costs for corn in Iowa to be between 32 and 174 USD/t CO_{2eq}. These are higher than the costs on the Typical Farm, which are negative. In the worst case of the Typical Farm, the costs are 74 USD/t CO_{2eq}, which is in the range of the estimation of the study. However, contrary to this thesis, the study by ICF assumes a yield reduction resulting from the reduced nitrogen rate. They calculate that foregone revenue from the reduced yield is higher than the reduction in costs from the reduced fertilizer rate, which explains the positive costs compared with the assumptions from the Typical Farm.

4.5.3 Reduction of tillage intensity (US-2)

As mentioned in chapter 4.4.1, the Typical Farm in the baseline scenario is operated with reduced tillage, which is the most common tillage system in Iowa. As presented in chapter 3.6.2.2, sequestering carbon in the soil can be used as a GHG mitigation strategy. This sequestration potential depends on the difference in the soil organic carbon content under no-till compared with reduced tillage.

A literature review on the soil organic carbon content of the average soils in Iowa under both tillage systems in the corn-soybean rotation reveals the following: Al-Kaisi et al. (2005) and Al-Kaisi and Yin (2005) assessed the effect of varying tillage intensity in Iowa. Both studies found a positive correlation between reducing tillage intensity and an increase of soil organic carbon, implying that no-till does promote retention of carbon. However, these studies were conducted over seven and three years, respectively. This period can be considered too short to determine the final carbon content of the system under the changed tillage systems, as stated by Al-Kaisi et al. (2005). Yet, they confirm the positive effect of no-till in the soil's carbon content in Iowa. Causarano et al. (2008) conducted a simulation for Iowa to estimate the effects of tillage in the soil organic carbon in a corn–soybean rotation. They provide estimates of the potential of the soil to capture carbon when transitioning between tillage systems. However, the author's descriptions of the operations conducted in each tillage system differ considerably from what is used in the Typical Farm[14]. Hence, none of the tillage systems provided in the study could be compared with the baseline scenario of the Typical Farm. Therefore, their results are deemed as not applicable for this analysis. No further pertinent literature could be identified. Consequently, the approach described in chapter 3.6.2.2 is utilized.

To estimate the carbon sequestration potential, it first is necessary to determine the soil type and the climate in the region. The soil is assumed to be a Mollisol, based on the map provided by Natural Resources Conservation Service (2021)[15]. The climate zone also is needed for this estimation, which was obtained using Sayre et al. (2020), as presented in chapter 4.5.1. With these data, it is possible to determine the reference carbon stock of the soil where the Typical Farm is located.

---

[14] The author's definition of reduced tillage implies the use of disk only in soybean and direct seeding in corn without any tillage operations. This differs considerably from the operations in the Typical Farm in the baseline scenario, which conducts tillage operations in both crops. Their description of conventional tillage also was deemed not applicable. Only no-till shared the same definition as in the Typical Farm.

[15] The equivalent of Mollisol in IPCC (2019a) definitions is High Activity Clay.

To determine the current soil organic content with the IPCC (2019a) methodology, it is necessary to adjust the reference carbon stock. This adjustment depends on the type of land use, the input of organic matter and the tillage intensity[16]. Each of these three elements has a coefficient that is multiplied with the reference stock carbon of the soil to calculate the estimated soil organic carbon content.

In the methodology by IPCC (2019a), the coefficients used for each type of tillage are provided with a default value and a range. This implies that the adjusted carbon stock of the soil under no-till also can be expressed as a range. The range provided by IPCC is a statistical calculation derived from the literature review. Given that the GHG mitigation potential of this strategy depends on the difference in the soil organic carbon content under reduced tillage and no-till, the range from IPCC can be used for the definition of the cases. Consequently, based on the coefficient of no-till, the default value is used for the standard case; the highest coefficient is used for the best-case scenario as it implies the highest mitigation potential. The lowest coefficient is used for the worst case, as it implies the lowest mitigation potential. Each of these cases is compared with the default coefficient of reduced tillage. A summary of all the coefficients used in this case as well as the resulting carbon stock of the system under reduced tillage and no-till is presented in see Table A.14 in the Appendix.

Following the IPCC methodology, carbon sequestration is a process that occurs linearly over a period of 20 years. After this period, the soil reaches a new equilibrium of soil organic carbon content, and it no longer can capture additional carbon from the atmosphere. The new stable level is maintained as long as the farm's tillage system is not changed. However, if the farm changes from no-till to a more intensive tillage system, the carbon sequestered will be released as CO_2. Hence, the GHG mitigation effect of this strategy is reversible (see chapter 3.5.6).

Table 4.8 Soil organic carbon stocks and sequestration potential for no-till vs. reduced till for Typical Farm in Iowa

Case	Soil organic carbon reduced tillage (t CO_{2eq}/ha)	Soil organic carbon no-till (t CO_{2eq}/ha)	Difference (t CO_{2eq}/ha)	Annual GHG mitigation potential (kg CO_{2eq}/ha)
Standard	216.4	227	10.4	520
Best	216.4	236	19.5	973
Worst	216.4	218	1.3	67

*Note: to transform CO_{2eq} to C, the CO_{2eq} value must be divided by 3.67.
Source: own estimation.

The carbon stock difference of each case and the annual carbon sequestration potential of each case are presented in Table 4.8. To attain the GHG mitigation potential of carbon sequestration in its entirety, it is necessary to operate the farm under no-till every year. Otherwise, only a share of the mitigation potential can be realized. In the case of the Typical Farm in Iowa, this implies conducting changes in both corn and soybeans. Consequently, the feasibility and impacts of changing to no-till must be assessed for each crop.

Transitioning the Typical Farm to no-till requires stopping the operations that disturb the soil, even at shallow depths, requiring changes in operations in corn and soybeans. In corn production, two operations must be stopped: utilizing the disk ripper and the pass with the soil finisher. In soybeans, just the soil finisher no longer can be used. Since this equipment is used only for these operations, transitioning to no-till implies

[16] The focus group stated that in corn–soybean rotations, crop residues are chopped and incorporated into the first few centimeters of the soil, leaving approximately 30% to 40% of the surface covered. This is in accordance with the definition of reduced tillage provided by IPCC for the calculation of carbon stocks.

it no longer is necessary. Hence, it is assumed that the grower no longer keeps them in the inventory, which reduces the cost of repairs, financing and depreciation entailed in this equipment. Since the passes no longer are conducted, less labor and diesel from the operations lead to a reduction in the costs compared with the baseline scenario. The costs involved in the repairs of maintenance of the tractor pulling the equipment are also reduced. However, the depreciation of the tractor does not change at the farm level.

Tillage operations that no longer are conducted have the objective to integrate part of the crop residues into the soil to facilitate decomposition. Without these, the corn stalks are left on the soil surface and can damage the tires of the seeder and fertilizer spreader. Consequently, the focus group agreed that it would be necessary to purchase stalk stompers. This addition bends the corn stalks to reduce the wear on tires and the risk of puncturing them. They estimate that the investment would cost 3,500 USD and would need to be replaced every five years with zero resell value.

Regarding the seeding of the crops in no-till, the members of the focus group stated that most modern planters used in reduced tillage systems also can be used in no-till without the need to reduce the speed of the tractor during seeding. However, they mention that it requires more equipment adjustment, especially the row cleaners. Hence, no changes to the seeding equipment are assumed.

Whether or not changing the tillage system from reduced to no-till affects corn yields depends on the soil and weather conditions. Al-Kaisi et al. (2015) conducted field trials in various soil types across Iowa between 2003 and 2013 and concluded that in soils with low levels of clay, under a no-till, corn yields do not statistically differ compared with reduced or conventional tillage. The same results are found by Al-Kaisi and Kwaw-Mensah (2007) and Al-Kaisi et al. (2005). However, according to Al-Kaisi et al. (2015), no-till performed worse in poorly drained soils, with yield reductions of 6% compared to the other tillage systems. Al-Kaisi and Yin (2004) come to the same conclusion regarding the impact of no-till on corn yields in heavy soils, indicating a reduction of approximately 5%. Additionally, Vetsch et al. (2007) report an approximately 1.8% reduction in no-till soybean yields compared with reduced till in a corn–soybean rotation over five years. The authors attribute this yield penalty from no-till mainly to the weather and soil conditions, especially water infiltration, as heavier soils tend to have problems with drainage and therefore are prone to waterlogging (Al-Kaisi et al., 2015; Licht and Al-Kaisi, 2005). Tillage operations loosen the soil particles and break them down, which facilitates water infiltration. Hence, without tillage, the flow of water in the soil may be limited, which causes the problems described.

A strategy that, in theory, could reduce the negative effects of no-till on yields is the installation of tile drainage in the fields. This practice consists of installing underground pipes with holes that allow the movement of water into them. Once in the pipe, the water can then flow into canals and leave the field. With this system installed, the excess water can be removed at a higher rate from the soil to reduce the risk of waterlogging. No scientific literature applicable for this region could be found to assess this measure. Researchers and experts from Iowa State University were contacted as well but could not confirm the effect beyond anecdotal evidence. Hence, while in technical terms it is possible to implement tile drainage, its effect on the reduction of the yield penalty cannot be confirmed and it therefore is not considered in this analysis.

Although by definition the Typical Farm has no exact location, it is assumed to portray a farm located in the northern part of the state. Information on the water infiltration rate of the soil without the use of tile drainage is made available by the Geospatial Laboratory for Soil Informatics (2019). Based on their data, the drainage in this region is categorized predominantly as poorly drained. Consequently, it is assumed that the Typical Farm operates in soils with poor drainage, indicating that a yield penalty would occur.

However, no-till can improve the soil's structure over time, which facilitates the flow of water through the layers, yet the build-up to obtain this benefit may take several years (Lal et al., 2007). Given the short-term consideration of the studies mentioned above, it is not possible to determine a statistically significant long-term trend, which can take decades (Al-Kaisi et al., 2015). In this regard, Dick et al. (1991) report the results of plots that have been under no-till for 25 years and indicate decreasing yield penalty in corn, signifying that the gap, in some cases, can be closed. On this matter, a participant of the focus group, who had already implemented no-till in some of the plots without tile drainage, commented it took approximately 10 to 15 years to notice an improved soil structure and that there were occurrences of waterlogging in fields in the first three to four years after transitioning. Hence, despite the decrease in yields, it is assumed that due to an improvement in the soil conditions over time, especially water infiltration rate, it is possible to return to the yield levels obtained before the transition to no-till.

Therefore, for the calculation of the GHG mitigation strategy and its costs, a transition stage is assumed for the Typical Farm. Given the poor soil drainage in the region, this stage assumes a 6% penalty in corn yields based on Al-Kaisi et al. (2015) and a 1.8% yield penalty for soybeans, based on Vetsch et al. (2007). These values apply in the first year after the implementation of no-till. Following the IPCC (2019a) guidelines regarding the changes in soil organic carbon stocks, it is assumed that it takes 20 years until the soil reaches an equilibrium. Therefore, the yield penalty is assumed to decrease linearly over the 20 years in which the carbon sequestration takes place. This period depicts the transition stage in the Typical Farm. The penalty is assumed to become zero in the 21st year, which also entails that the carbon sequestration potential is realized.

The yield penalty of each crop implies a foregone revenue in the future. This economic loss must be discounted to express its current economic value. The following formula based on Mußhoff and Hirschauer (2020) can be used to calculate the total foregone revenue accumulated in the 20-year transition stage (Equation 31).

$$\boldsymbol{FR = \sum_{t=1}^{20} FY_t \times P \times (1+i)^{-t}} \qquad (31)$$

FR:	total foregone revenue of the crop in the transition stage (USD)
t:	year of transition
FY_t:	foregone yield for the crop in year t (t/ha)
P:	average crop price (USD/t)
i:	interest rate (%)

For each crop, the foregone yield is obtained by multiplying the yield in the baseline scenario with the calculated annual yield loss, as previously explained. The average prices of corn and soybeans for the period 2016 to 2018 are used, which are 131 USD/t and 339 USD/t, respectively. These are obtained from the focus group. An annual interest rate of 2% is assumed for the discounting as that is the value reported for the Typical Farm for long-term loans.

The total foregone revenue must be expressed as an annuity to divide the economic loss equally for each year of the transition stage. Based on Mußhoff and Hirschauer (2020), Equation 32 is used:

$$\boldsymbol{A = \frac{FR}{\left(\frac{(1+i)^{20}-1}{(1+i)^{20}\times i}\right)}} \qquad (32)$$

A:	annuity of the foregone revenue of the crop (USD/ha)

The annuity of the foregone revenue caused by the yield penalty is estimated to be 56 USD/ha in corn and 13 USD/ha in soybeans. Hence, the effect on an average hectare of the Typical Farm is 34 USD/ha, which is taken as an additional cost. Table B.1 in the Appendix depicts the year-by-year effect of the foregone revenue calculated.

The adjustment factor used to modify indirect GHG emissions depends on the yield, as this determines the nitrogen removal via harvest. Thus, the average yield penalty in the transition stage is deducted from the yield in the baseline scenario.

Estimation of emissions from Indirect Land Use Change (ILUC)

The yield penalty in the Typical Farm likely would generate ILUC, especially considering the relevance of Iowa in global crop production. ILUC occurs when the supply of a specific product is reduced or diverted while the demand remains unchanged. Given the comparatively lower supply, the price of the product would increase, which would encourage producers elsewhere in the world to increase their output to meet demand. They achieve this by intensifying their production on their existing acreage and/or by transforming land for agricultural use (Wicke et al., 2012). The intensification of the production signifies that higher rate of inputs are used, which in turn increases the GHG emissions per hectare. The transformation of land involves the release of GHG from land use change; that is, the carbon stored in the soil is released. Additionally, transforming the land could involve changes in the vegetation - for instance, from clearing woodland - which result in further GHG emissions.

To calculate the GHG emissions from ILUC, the values provided by the Environmental Protection Agency EPA (2010) are used. These values are crop-type-specific GHG emission values and were calculated to determine the environmental impact of the United States' biofuel policy "Renewable Fuel Standard Program." In their estimation of GHG emissions, they include the changes in the emissions from the intensification of production as well as land use change. Both elements are assessed at national and international level. There is a large degree of variation in the estimation of emissions from ILUC (Wicke et al., 2012). However, EPA's estimations are deemed to represent an average value within the spectrum of estimations.

The emissions derived from land use change used in the assessment from EPA (2010) are distributed equally over 30 years. To adapt those values to the 20-year transition stage assumed in this assessment, the EPA's values are multiplied by 3/2. This distributes the same total emissions from land use change over a shorter time[17]. The values used for the estimation of ILUC per weight of crop are shown in Table B.2 in the Appendix. It is estimated that the emissions from corn are 0.25 kg CO_{2eq}/kg crop and for soybeans, 0.35 kg CO_{2eq}/kg crop. These values are multiplied with the annual yield penalty and then the average for the 20 years is calculated. This average is considered in the category "land use." Hence, it is estimated that the average yearly yield penalty for corn causes 101 kg CO_{2eq}/ha due to ILUC. Soybeans' emissions are calculated to be 13 kg CO_{2eq}/ha. Thus, the average hectare (50% corn and 50% soybeans) causes additional emissions of 57 kg CO_{2eq}/ha. The annual emissions from ILUC from corn and soybeans are presented in Table B.3 in the Appendix.

Contrary to the approach from this assessment, EPA's GHG emissions from the intensification does not take into account carbon released from the manufacture of fertilizer. This would increase the ILUC emissions attributed to each crop. Nevertheless, considering that the emissions from manufacture are only a share of the emissions from intensification and, in turn, intensification represents only a share of ILUC, no adjustment is applied to the estimated values for corn and soybean.

---

[17] Since the values provided by EPA are used to compare fuel with biofuels, the GHG emissions are expressed as a function of energy; that is, kg CO_{2eq}/MJ. To transform this unit into kg CO_{2eq}/kg of crop, the amount of biodiesel obtained from soybeans and bioethanol gained from corn are needed. The values provided by Canter et al. (2016) are used for corn's bioethanol yield and the values from Chen et al. (2018) for soybeans' biodiesel yield. Both of these studies are conducted using average results from the mills in the US. Standard values on the biofuels energetic content as well as density are obtained from BioGrace, a GHG estimation tool from the EU (2015).

Theoretically, including emissions from ILUC in the transitory yield reduction could imply that carbon sequestration could occur. Carbon sequestration would take place in the regions where land was transformed for agricultural use to cover the gap in the supply. With the yield penalty no longer applying, this gap would be filled. Hence, the land could be returned to the use it had before the yield penalty appeared. With this return, the carbon released from the biomass and soil would be sequestered back. However, estimating whether the land is returned to its original use and the share returned implies changing the assumptions originally made by EPA (2010). Furthermore, it can be considered unlikely that the land is returned to its original state, as transforming the land involves costs for the grower. Using only the share of intensification entailed in EPA's estimates underestimate the impact of ILUC. Consequently, for the estimation of this mitigation strategy, EPA's values are used as previously estimated without further changes, since it is assumed that:

(1) The land transformed as a response to the ILUC from the yield penalty remains as agricultural land, hence, no sequestration occurs.

(2) The disappearance of the yield penalty in the Typical Farm implies that the growers would reduce the intensity of their production to the same level as before the yield penalty. This would be a response to the decrease in the crop prices they would perceive since the yield penalty no longer would be active and the demand would stay unchanged.

Total costs in transition stage from implementing the mitigation strategy

The changes in the cost of changing the Typical Farm from reduced tillage to no-till during the transition stage of 20 years are presented in Table 4.9. Only the categories that have differences in the costs are presented; the rest is not shown as it does not affect the resulting costs. The changes in costs between the cases do not differ, as all strategies imply the same adjustment of operations. Hence, only the results from the standard case are shown. Given that the change implies different effects in corn and in soybeans, it is necessary to estimate them independently. In corn, the cost reductions in terms of diesel, machinery and labor are higher than in soybeans. This is a consequence of corn having more tillage operations under reduced tillage. The reduction of these three elements combined totals 44 USD/ha in corn and 16 USD/ha in soybeans. However, the yield penalty depicted in the category "foregone revenue" also is significantly larger in corn than in soybeans (55 and 13 USD/ha, respectively). Consequently, the total cost from changing to no-till results in a cost increase of 11 USD/ha in corn but a decrease of 3 USD/ha in soybeans.

The category "remaining" entails the costs from the categories seed, fertilizer, pesticides, liming and contractor operations. None of these elements has costs in the baseline scenario and they are not affected by the implementation of the mitigation strategy. Consequently, the category "remaining" results in zero.

To assess the cost of the implementation of no-till per hectare, the effect of each crop in the crop rotation must be averaged. Given that the crop rotation is composed of two crops with equal acreage, a simple average is calculated. On average, costs would increase by 4 USD/ha during the transition period. This is the value used to determine the cost of the mitigation strategy in the Typical Farm.

Table 4.9 Difference in costs from no-till vs. baseline scenario for Typical Farm in Iowa in transition stage (USD/ha)

	Standard case		
	Corn	**Soybean**	**Average**
Diesel	-12	-4	-8
Labor	-8	-2	-5
Machinery	-24	-10	-17
Foregone revenue	55	13	34
Remaining	0	0	0
Total	**11**	**-3**	**4**

Source: own estimation.

Emissions per hectare in transition stage from implementing the mitigation strategy

Table 4.10 presents the change in the GHG emissions from transitioning the Typical Farm from reduced tillage to no-till on an average hectare. Only the categories that have differences in the costs are presented; the rest are not shown as they do not affect the resulting costs. On average, the release of emissions from the category "direct" would be 5 kg CO_{2eq}/ha lower compared with the baseline scenario. This category is only N_2O, and its reduction is a result of the average yield penalty. Lower yields imply that fewer crop residues are left after the harvest. Bacteria decompose these residues and release N_2O in the process. Given that the yields in no-till in the transition period are lower than those of reduced tillage, the emissions from crop residues also are reduced.

However, the yield penalty also implies that less nitrogen is removed via harvest. In corn, the nitrogen fertilization rate is unchanged compared with the baseline scenario. Hence, more nitrogen is susceptible to leaching and volatilization, which are the elements entailed in the category "indirect". Therefore, the emissions from this category are 12 kg CO_{2eq}/ha higher than in the baseline. The same values apply to all cases. The emissions from diesel are reduced by 41 kg CO_{2eq}/ha as the passes for tillage in the baseline scenario no longer are conducted.

The category "remaining" in this case entails the emissions from urea, liming and manufacture. While emissions in the category "manufacture" in the baseline scenario and with the mitigation strategy are different than zero, they are the same in both cases as the mitigation strategy does not affect the fertilization rate. Urea and liming emissions both are zero with and without the strategy.

The largest share of the reduction of emissions comes from the category "land use," which entails the carbon sequestration. Sequestration is realized only for the 20 years of the transition period. Potential carbon sequestration varies across the cases based on the values from IPCC. This results in a mitigation potential in the average hectare of 463 kg CO_{2eq}/ha in the standard case, 916 kg CO_{2eq}/ha in the best case, and only 10 kg CO_{2eq}/ha in the worst case. In all these cases, the average emissions from ILUC are discounted. These emissions reduce the mitigation effect from land use by 57 kg CO_{2eq}/ha.

In the worst case, the effect of ILUC makes the emissions from corn in the category "land use" positive. This implies that the emissions from ILUC would be higher than the carbon sequestration occurring in the Typical Farm. However, the net effect of implementing no-till in corn would still reduce GHG emissions compared with the baseline.

As in the case with costs, it is necessary to consider an average hectare for the estimation of GHG emissions. This average is calculated following the same procedure as for costs. In the standard case, GHG emissions would be reduced on average by 497 kg CO_{2eq}/ha, by 950 kg CO_{2eq}/ha in the best case and by 44 kg CO_{2eq}/ha in the worst case.

Table 4.10 Difference in GHG emissions no-till vs. baseline scenario for Typical Farm in Iowa in transition stage (kg CO_{2eq}/ha)

	Standard case			Best case			Worst case		
	Corn	**Soybean**	**Avg.**	**Corn**	**Soybean**	**Avg.**	**Corn**	**Soybean**	**Avg.**
Direct	-9	-1	-5	-9	-1	-5	-9	-1	-5
Indirect	24	0	12	24	0	12	24	0	12
Diesel	-61	-20	-41	-61	-20	-41	-61	-20	-41
Land use	-419	-507	-463	-872	-960	-916	34	-54	-10
Remaining	0	0	0	0	0	0	0	0	0
Total	**-465**	**-529**	**-497**	**-918**	**-982**	**-950**	**-12**	**-76**	**-44**

Source: own estimation.

Emissions per ton of product in transition stage from implementing the mitigation strategy

Evaluating the emissions per ton of product shows whether the net output of emissions would increase. This would occur if the mitigation strategy results in a net increase of emissions per ton of crop harvested, as the demand for the product can be assumed to remain constant.

The changes in the GHG emissions in corn and soybeans derived from transitioning to no-till are presented Table 4.11. For this calculation, the emissions per hectare are divided by the yield of the respective crop. In the case of the mitigation strategy, the yield used in the division is reduced by the average yield penalty of the 20 years. For each crop, the standard, best and worst cases are calculated using the same reduced yield.

For all cases, the GHG emissions per ton of product in the transition stage are lower than in the baseline. Consequently, the average hectare has a net reduction of 85 kg CO_{2eq}/t of product. However, the total negative effect is mainly a result of the reduction obtained with land use, which represents carbon sequestration. There is a reduction in the emissions from diesel, yet this is a small share of the reduction compared with land use.

Nevertheless, considering yield penalty, the reduction in emissions presented per hectare and per ton of product reveal additional information. Per ton of product, the change in GHG emissions from the category "direct" has a positive value. This differs from the result per hectare presented in Table 4.10 for the same category, which is negative. This difference is a result of the yields in no-till being lower in the transition stage and the nitrogen application rate remaining unchanged. A share of the GHG emissions from direct is determined by the crop residues, which are less due to a lower yield, reducing the emissions. However, the emissions from nitrogen fertilization entailed in direct are the same as in the baseline. This positive value implies that, while fewer emissions are triggered by crop residues, this is not enough to compensate for the same emissions from nitrogen fertilization being distributed over less yield.

The same situation occurs in the case of the emissions from manufacture of fertilizer. This category depends on the amount and type of fertilizers applied. Given that the same rate is used with no-till but the yield is lower than with reduced tillage, the same GHG emissions are distributed over less yield. Manufacture is under the category "remaining" in Table 4.10, as it has no change on a per-hectare basis.

Compared with the emissions savings from land use, the increase in the emissions from these categories can be considered low.

Table 4.11 Difference in GHG emissions no-till vs. baseline scenario for Typical Farm in Iowa in transition stage (kg CO_{2eq}/t)

	Standard case			Best case			Worst case		
	Corn	Soybean	Avg.	Corn	Soybean	Avg.	Corn	Soybean	Avg.
Direct	4	0	2	4	0	2	4	0	2
Indirect	2	0	1	2	0	1	2	0	1
Manufac.	2	0	1	2	0	1	2	0	1
Diesel	-4	-5	-5	-4	-5	-5	-4	-5	-5
Land use	-34	-133	-84	-71	-252	-162	3	-14	-6
Remaining	0	0	0	0	0	0	0	0	0
Total	**-31**	**-139**	**-85**	**-68**	**-258**	**-163**	**6**	**-20**	**-7**

Source: own estimation.

Mitigation costs in transition stage

As a result of transitioning the farm from reduced tillage to no-till, over the period of 20 years considered the transition stage, the cost to mitigate GHG would be 8 USD/t CO_{2eq} in the standard case, 4 USD/t CO_{2eq} in the best case and 89 USD/t CO_{2eq} in the worst case. The considerations on the adoption of this strategy are discussed at the end of this chapter.

Total costs in stable stage from implementing the mitigation strategy

To maintain the carbon sequestered in the soil and avoid its release as GHG, the Typical Farm needs to stay under a no-till system. Hence, it is necessary to assess the costs in the stable stage after the transition period. The changes in the costs of the different cases are the same as in the transition stage (see Table 4.9) except for the foregone revenue. After the transition stage, it is assumed that yields are back to the level that they were in the baseline scenario. Hence, the foregone revenue in the stable stage becomes zero. Without it, the costs in the three cases become the same, as only this category was different across the three cases.

Given that no changes in terms of operations occur after the transition stage, the same reduced expenditure in machinery, labor and diesel realized in the transition stage is valid in the stable stage. This applies to all cases and both crops.

The resulting reductions in costs in the stable stage from transitioning to no-till is 44 USD/ha for corn and 16 USD/ha for soybeans, with an average reduction of 30 USD/ha. In theory, this is an indication that the grower would tend to maintain no-till and not revert to reduced tillage, as no-till is estimated to be more profitable. Therefore, based on the current situation, there are no perceivable risks that the grower would return to its previous tillage.

As presented in chapter 4.5.1, the costs estimated for corn in the baseline scenario are 1,008 USD/ha. The cost reduction obtained after the 20-year period for this crop represents a 4.4% savings, which can be considered marginal. Furthermore, the prospect of having a foregone revenue over 20 years may deter the grower. To illustrate, the reduced yield of corn in the first year would imply a foregone revenue of 98 USD/ha, as shown in Table B.1. The total cost to implement no-till in corn could be seen as 908 USD/ha, which is the total foregone revenue from all years combined considering the discounting.

Emissions per hectare in stable stage from implementing the mitigation strategy

The emissions from no-till in the stable stage are the same as in the baseline scenario in all categories except for diesel. This is because the farm operations conducted in the transition stage are maintained in the stable stage. Consequently, the same values from diesel as in the transition stage (see Table 4.10) are valid for the stable stage. Given that no other categories differ, the change in emissions from diesel represents the total change in emissions per hectare in the stable stage. In corn, this decrease is 61 kg CO_{2eq} /ha and in soybeans, 20 kg CO_{2eq} /ha. The average hectare has a reduction of 41 kg CO_{2eq}/ha in the stable stage. These results apply to the standard, best and worst cases of the mitigation strategy.

The carbon sequestration potential is assumed to be fulfilled in the stable stage. Hence, the soil no longer binds additional carbon from the atmosphere. Consequently, compared with the transition stage, the category "land use" becomes zero. The emissions from ILUC also are assumed to be zero since the yield penalty is overcome.

Mitigation costs in stable stage and considerations on the adoption of the strategy

Since the resulting GHG emission in the stable stage are lower than in the baseline, the Typical Farm would continue to reduce emissions. The changes in costs in the stable stage result in a reduction of 30 USD/ha compared with the baseline scenario. Consequently, a GHG mitigation cost can be calculated for the stable stage. For all cases, the reduction of emissions in the stable stage would have a cost of -739 USD/t CO_{2eq}. This negative cost indicates that the stable stage poses a win–win situation, which is discussed in chapter 8. This means that growers using the strategy not only would reduce the GHG from crop production but also would reduce their costs. The large number is a result of the comparatively small GHG savings, which result only from less diesel usage.

The costs during the stable stage can be interpreted as the annual expenditure to avoid growers reverting to reduced tillage. Reverting would release the carbon sequestered in the soil in the transition stage. However, as already mentioned, the negative cost in the stable stage implies that it is unlikely that the grower transitions back to reduced till, unless the conditions are changed.

This strategy could be implemented on a wide scale in a short time frame, as it does not require a significant investment. The only additional expenditure is the corn stalk stampers, which have a cost that was considered negligible at the farm level. Nevertheless, the implication of having a yield penalty can dissuade growers from transitioning to no-till. Using the averaged foregone revenue (34 USD/ha) may be misleading, as this value varies considerably. The actual foregone revenue on the average hectare in the first year after the transition is 60 USD/ha. This indicates that the total costs on the average hectare in year one is 30 USD/ha higher than in the baseline scenario. This prospect can discourage growers from adopting the strategy. In addition, the learning curve associated with switching the tillage system acts as another deterrent (Al-Kaisi et al., 2015).

Furthermore, the strong dependency of no-till on chemical control for weeds also can have a negative effect on the likeliness of adoption of this strategy, as many weed species are developing resistance toward herbicides (Al-Kaisi and Yin, 2004). Without tillage operations to limit the weeds' growth, no-till could be expected to have reduced yields from weed pressure in the future. Alternatively, the growers may need to use a different combination of herbicides, which may increase their expenditure. The possibility to implement additional extension programs to help growers overcome the intricacies of the switching to no-till along with the possibility to compensate for the forgone yield during the transition period could be suited strategies to promote the transition. As with the optimization of nitrogen fertilization in corn (see chapter 4.5.2), this strategy could be implemented stepwise, meaning that only a fraction of the acreage is managed under no-till. This would lower the perceived risk by the grower.

The monitoring and control of this strategy could be conducted on a state-wide level using satellite images. This strategy implies leaving the crop residues on the soil surface, as opposed to reduced tillage, which incorporates a share into the soil. Hence, the difference between the systems is visible. Consequently, scanning the field with satellites after the harvest and before the next crop is growing would reveal whether no-till is being used. This approach would lower the necessity to conduct farm visits, which would reduce the cost of monitoring and enforcing the GHG mitigation strategy.

Comparison of results from Typical Farm with literature

McNunn et al. (2020) simulated the GHG mitigation potential of reducing tillage intensity to no-till to be 1,477 kg CO_{2eq}/ha. This value is the average result from the strategy being implemented in corn and soybeans, which is considered comparable to the average hectare used in the Typical Farm. However, the potential is significantly higher than even the best case of the Typical Farm in the transition stage (950 kg CO_{2eq}/ha for the average hectare). Contrary to the Typical Farm, which assumes a reduced tillage regime in the baseline, their depiction of the baseline is based on a farm with conventional tillage. The carbon sequestration potential from a transition from conventional to no-till is higher than from reduced tillage to no-till (IPCC, 2019a). Consequently, the difference in the baseline largely explains the variance in the results.

ICF (2013) estimates the mitigation potential to be 420 kg CO_{2eq}/ha for corn. This is similar to the estimation in the Typical Farm in the transition stage in the standard case, which is 465 kg CO_{2eq}/ha. For soybeans, the authors estimate an abatement of 130 kg CO_{2eq}/ha, which is between the standard and worst cases of the Typical Farm (529 and 76 kg CO_{2eq}/ha, respectively). The estimations from these papers do not account for changes in the emissions from diesel from the reduced number of operations. Their inclusion would increase the mitigation potential calculated in the papers. To illustrate, the reduction in emissions from the lowered diesel usage in corn in the Typical Farm amounts to 61 kg CO_{2eq}/ha. Adding this value to the calculation in the ICF (2013) study would increase the reduction to 481 kg CO_{2eq}/ha, reducing the difference compared with the Typical Farm.

The mitigation costs provided by ICF (2013) are 30 USD/t CO_{2eq} for corn and 77 USD/t CO_{2eq} for soybeans. The corresponding costs in the standard cases of the Typical Farm are 23 USD/t CO_{2eq} and 6 USD/t CO_{2eq}, correspondingly. Thus, while the costs in corn can be considered comparable, the difference for soybeans is significant. Nonetheless, in both cases, the difference can be attributed to the assumptions on input costs and emissions considered in the overall analysis.

4.5.4 Cover crops (US-3)

The Typical Farm does not use cover crops in winter. This practice is not common in Iowa, as it is reported to be used on less than 5% of the acreage, based on data from 2017 (USDA, 2020). Cover crops can be used to reduce the leakage of nutrients, especially nitrogen, during the winter as they uptake them and use them for their development (Thapa et al., 2018). These nutrients are returned to the soil after the cover crop is terminated and incorporated into the soil as green manure. Furthermore, legumes, which not only uptake nutrients but also can fixate nitrogen from the air to use for their development, can be used as cover crops as well. Bacteria fixate nitrogen to the soil which can be absorbed by the following crop. Moreover, in a process called mineralization, this fixated nitrogen also is released when the cover crop residues decompose and can be used by the following crop. This nitrogen can be considered as a credit toward the next crop and can be deducted from the nitrogen fertilization plan. Therefore, as explained in chapter 3.6.2.3, implementing this practice offers the possibility to reduce GHG emissions by reducing the need to use synthetic fertilizers, lowering the emissions attributed to its manufacture as well as the emissions the fertilizer triggers in the soil.

Crop residues from the cover trigger N_2O emissions, but this effect is proportionately small compared with the potential emissions saving from reducing the fertilizer rate. Furthermore, the cover crop's biomass can be incorporated into the soil, which leads to carbon sequestration. The sequestration of carbon is considered a GHG mitigation, as it binds carbon from the atmosphere to particles in the soil. To achieve the full potential of carbon sequestration, it is necessary to use winter cover crops in the entire crop rotation. Otherwise, only a fraction of the carbon sequestration potential is realized. A partial implementation of this strategy implies that less biomass incorporated into the soil and a higher loss of carbon in the periods without a crop. This means that a cover crop must be used after soybeans as well as after corn. The feasibility and considerations for each crop must be assessed individually.

To implement winter cover crops, the timeframe considered is between the harvest of one crop and the seeding of the next one. In the Typical Farm, the harvest of soybeans usually occurs in late September and is followed by the harvest of corn in early October. The seeding of the following crops takes place from mid- to end of April. Furthermore, considering the winter conditions in Iowa (see chapter 4.5.1), to have a successful establishment of the cover crop, it is necessary to use plant species that start growing shortly after seeding and can survive the winter. Plants that can survive the winter are referred to as winter hardy. Without this characteristic, the effect of avoiding leakage and fixing nitrogen is not attainable as the cover crop survives until spring.

Given that soybeans can fixate nitrogen and do not require external nitrogen inputs, having a legume as a winter cover crop before soybeans may increase the risk of leaching. This can occur because the soybeans may not take up the nitrogen fixated by the cover crop. Therefore, to achieve the carbon sequestration potential and minimize the risk of leaching, a non-legume winter cover crop is used before soybeans. For the winter cover crop coming after soybeans and before corn, a legume can be chosen, as soybeans do not generate a nitrogen credit for the following crop (see chapter 4.5.2). With this approach, the carbon sequestration potential can be realized and the emissions from nitrogen fertilization in corn can be reduced.

Taking these considerations into account, the focus group mentioned that cereal rye was the cover crop that they had successfully tested before soybeans. As for the legume cover crop before corn, they stated that hairy vetch could be established. Although they had tested red clover as an alternative as well, they mentioned that it was not possible to obtain a uniform cover as not all the seeds emerged. Hence, the estimation of the mitigation strategy is conducted for cereal rye and hairy vetch. The seed rate stated by the focus group for these cover crops is 84 kg/ha and 34 kg/ha, respectively.

The seed prices mentioned by the focus group are 0.5 USD/kg for cereal rye and 4.4 USD/kg for hairy vetch. However, they also stated there is no competitive market for seeds for winter cover crops, as this practice is not common in the area. Arguably, this implies that the current seed prices for vetch and cereal rye may be higher than if the practice was more used. Theoretically, if cover cropping becomes mandatory, growers and seed producers would use the following season to produce seeds for cover crops. The additional seed production would quickly increase the supply, lowering the price in a short time frame. Consequently, estimating this GHG mitigation strategy using current seed prices would depict the mitigation cost only for the short-term. Hence, to calculate GHG mitigation costs that represent a long-term scenario, the theoretical seed prices are estimated.

An alternative for growers using rye as a winter cover crop would be to utilize farm-produced seeds. Since cover crops are not harvested, the grower does not need to purchase certified seeds to maximize grain yields. Hence, the rye produced by other farms would suffice to establish the cover crop. If this were the case, the grower producing rye as a cash crop would accept the market price for the product, as his decision is not affected by the buyer's intended use. Therefore, it can be assumed that the market price of rye is an indication of the cost that seeds for cover crops would have in the long-term. Therefore, a seed cost of 0.2 USD/kg is assumed for rye, based on the price from 2016-2018 (USDA, 2021b).

The same principle applies for hairy vetch. However, no information on the average price of this crop in the USA or at the international level could be found. Hence, prices are obtained from consultation with experts of the *agri benchmark* Cash Crop network[18]. Thus, a price of 0.4 USD/kg is assumed.

An additional 8 USD/ha is assumed for the inoculant for hairy vetch. The inoculant, in this case, *Rhizobium* is the bacteria responsible for the biological fixation of the nitrogen in the soil. Without it, the nitrogen fixation potential is severely limited, which not only reduces the growth rate of the cover crop but also reduces the potential nitrogen available for the following crop (Lawson et al., 2013).

Although seed prices have been adjusted for this analysis, the presumptions used in the adjustment are based on the current market situation, which has a comparatively low demand for this type of seed. Thus, it is possible that in a scenario with cover cropping becoming mandatory, the widespread demand could lower the prices for these seeds beyond what has been assumed here, as more seed producers would enter the market, increasing supply. Nonetheless, given that no price projections exist for these crops, the prices assumed for the calculation are deemed a valid approximation.

Considering the seeding rates and assumed costs for seeds, rye is calculated to cost 18 USD/ha and vetch, 22 USD/ha. Given that the cover crops in the Typical Farm are terminated before they reach maturity, it is not possible to save seeds for later use. No fertilizers or pesticides are applied to the cover crops during their growing time.

In Iowa, the seeding of cover crops typically is conducted by contractors, according to the focus group. The average price for this service is 42 USD/ha. In this price, it is assumed that both cover crops are seeded using a seed drill. The alternatives mentioned in the discussion were using an airplane or broadcasting. However, both were considered inadequate due to the lack of uniformity in the distribution of the seeds. Hence, despite contractors offering these alternatives at lower rates per acre compared with drill seeders, the need to use more seed and the risk of not establishing a proper plant population deter growers. Consequently, the cover crop is assumed to be drilled by a contractor directly after the harvest of the crops. The diesel usage of the contractor (3 l/ha) is included in the GHG emissions.

The nitrogen fixation potential of legumes is largely dependent on the growing period and weather conditions. The focus group mentioned that they estimated the nitrogen biological fixation potential of hairy vetch in Iowa to be between 28 and 39.2 kg N/ha. To assess this statement, a literature review was conducted, considering the crop rotation, harvest and seeding dates and the climate in Iowa.

Ruffo and Bollero (2003) and Bollero and Bullock (1994) calculated the fixation potential of hairy vetch in northern Illinois. The test fields of the studies are in a similar climatic condition, but the reported winter average temperatures are approximately 1° to 2°C higher than in the case study region in Iowa. The authors of both studies concluded that nitrogen credit from vetch for the next crop is approximately 40 to 45 kg N/ha. These studies were conducted using a seeding rate similar to the one stated by the focus group. Given the slightly lower temperatures in Iowa in winter, the growth rate of vetch would be slower and it would fixate less nitrogen (Brainard et al., 2012). Consequently, for the nitrogen fixation of vetch in Iowa, the average of the range (34 kg N/ha) provided by the focus group is used as the standard case for the estimation. This represents a reduction of the nitrogen fixation of 23% compared with the literature. The bottom and upper values of the range the focus group stated (28 and 39 kg N/ha) are used for the best and worst case analyses. The rate of anhydrous ammonia is reduced by the nitrogen fixated. Moreover, this assumption also implies that the nitrogen available in the system used for the estimation of the adjustment factor remains unchanged compared with the baseline scenario. The estimation on the nitrogen fertilization of corn for each case is shown in Table 4.12.

---

[18] Interview with agricultural analyst of the Spanish National Network of Typical Farms (personal communication, 30.05.2021)

Table 4.12 Nitrogen rate and supply from fixation by cover crop for Typical Farm in Iowa

Case	Total nitrogen (kg N/ha)	Synthetic nitrogen (kg N/ha)	Nitrogen from fixation (kg N/ha)
Standard	181	147	34
Best	181	142	39
Worst	181	153	28

Source: own estimation.

As discussed in chapter 3.6.2.3, winter cover crops have been shown to reduce the NO_3 leaching rate from nitrogen fertilizer into groundwater (Meisinger, 1991; Tully and Ryals, 2017). However, considering that hairy vetch is seeded after soybeans, which are assumed to not generate a nitrogen credit for corn, the nitrogen that could be retained is considered as limited. Hence, it is assumed that hairy vetch does not alter NO_3 losses. Rye is used before soybeans, which do not receive nitrogen fertilizer in the fall, which is among the key drivers of leaching (Di and Cameron, 2002). This cover crop could retain a share of the nitrogen left in the soil after the harvest of corn. Field trials in Iowa indicate that the reduction of nitrogen leaching by rye in corn–soybean rotations is less than 4 kg N/ha per season (Zhiming Qi et al., 2008). Moreover, these trials terminated the cover crop later than what is assumed in this assessment. Therefore, the trials had a longer time window in which leaching was reduced by the cover crop. Consequently, the reduction of leaching for the Typical Farm is likely to be even smaller. Thus, it is assumed that rye's effect on leaching is negligible in this case.

It is possible to delay the seeding of corn to enable the cover crop to have a longer growth period in spring, providing more time to fixate more nitrogen. However, considering the fixation rate, it is recommended to terminate the cover crop to maintain the seeding date of corn as delaying it would reduce corn yields (Ruffo and Bollero, 2003). Hence, the standard planting date of corn is maintained, as the foregone revenue from delaying seeding is higher than the savings in fertilizer attainable from an extended nitrogen fixation period of vetch. To illustrate, based on results provided by Abendroth et al. (2017), delaying the seeding of corn by two weeks could imply a revenue loss from the reduced yield equivalent to 45 kg N/ha as anhydrous ammonia. This is higher than the estimated nitrogen fixation potential of vetch on the Typical Farm.

No negative effects on corn yield are reported from growing a cover crop and the residues of hairy vetch are indicated to be completely decomposed by the end of corn's growing season (Ruffo and Bollero, 2003). Similarly, rye crop residues are reported to not generate negative effects on the soybean yields, based on field trials conducted in Iowa (Moore et al., 2014).

In the case of the Typical Farm, it is assumed that both cover crops are terminated with glyphosate in early spring. This implies an additional pass with pesticide in corn and soybeans, which increases costs of diesel and labor, as well repairs and maintenance of the pesticide sprayer. In the current tillage program of the farm, superficial tillage is conducted before seeding. This process integrates the residues of the cover crops into the upper layers of the soil to promote its decomposition. This lowers the risks of negatively affecting the development of the corn and soybean seedlings due to the cover crop's biomass (Wayman et al., 2015). Consequently, no changes in yields are assumed.

For the estimation of the GHG emissions, the biomass of crops grown is considered a source of N_2O emissions. This includes cover crops and is caused by the biomass of the plants acting as a substrate for the bacteria to process. The processing by the bacteria transforms a share of the nitrogen in the crop residues into N_2O. Consequently, if more biomass is available, more emissions will be released. For this calculation, the biomass of hairy vetch reported by Ruffo and Bollero (2003) and Bollero and Bullock (1994) is used as a reference, as these studies are also considered for the assessment of the nitrogen fixation potential already presented. On average for all locations in both studies, the authors reported a yield of 1.4 t/ha as dry matter. However, taking into account the considerations of the colder winter in Iowa limiting the growth potential of the vetch, the biomass is adjusted as well. Thus, the dry matter from the studies is reduced using the same percentage difference assumed for nitrogen fixation, which was previously estimated as 23%. Therefore, the dry mass of hairy vetch is reduced to 1.1 t/ha. For the biomass of rye, the results from Moore et al. (2014) are used, which indicate a dry mass of 2.7 t/ha without the use of fertilizers and a seeding density similar to that reported by the focus group. Given that these results are from Iowa and for similar harvesting and seeding dates as those in the Typical Farms, no adjustment is made[19].

No scientific literature could be identified that evaluates the long-term effect of cover cropping in soil organic carbon in conditions similar to the Typical Farm. Still, research by Kaspar et al. (2006) can be used to provide a reference. In this three-year project in Iowa, the use of rye as a cover crop following soybeans in rotation with corn was tested. They concluded that rye had a positive effect on the soil organic carbon, implying that carbon sequestration took place. However, in another study conducted in eastern South Dakota, a state located to the northwest of Iowa, Chalise et al. (2019) could not identify a significant effect in carbon sequestration. In this three-year study, the authors tested a mixture of hairy vetch and rye in combination in a corn–soybean rotation.

Nevertheless, the effect of cover crops on soil organic carbon is not always detectable in the first year of its implementation (Acuña and Villamil, 2014; Blanco-Canqui et al., 2014). Therefore, to properly determine the effect of cover crops on soil organic carbon, long-term studies are necessary. Consequently, in the absence of location-specific data from long-term research, the methodology provided by IPCC (2019a) is used for the estimation of the carbon sequestration potential (see chapter 3.5.6).

To use this approach, it is necessary to consider the soil and climate of the region to determine a reference carbon stock of the soil. This estimation was already conducted for the reduction of tillage intensity (US-2). As in that case, to estimate the current carbon stock of the soil, the reference value is adjusted using coefficients for the type of land use, the input of organic matter and the tillage intensity. The use of cover crops is included in IPCC's coefficient for the input of organic matter. IPCC provides these factors as a default value with a range. The default value is used for the standard case. The upper value of this multiplication factor is used for the best case as it implies the highest soil organic carbon. Correspondingly, the bottom value is used for the worst case. IPCC assumes for the worst-case scenario that the cover crops do not generate carbon sequestration. Therefore, in the worst-case scenario, the GHG mitigation effect of carbon sequestration is assumed to be zero.

IPCC's coefficients are a statistical inference derived from the distribution of results obtained from a literature review. In the case of the coefficient for the input of organic matter, it is not specific for the practice of cover cropping. This factor aggregates other practices aimed at increasing the production of biomass on the field to promote carbon sequestration. Therefore, the result of the worst case not sequestering carbon may be a consequence of other practices not affecting soil organic carbon. Poeplau and Don (2015) conducted a meta-analysis focused on assessing the effect of winter cover crops on soil organic carbon. They found a statistically significant correlation between this practice and carbon sequestration. It is not possible to transform their results to match IPCC's coefficients. Consequently, following IPCC's methodology, the worst-case scenario considers no carbon sequestration in the soil, yet the probability of this being the case is considered highly unlikely.

---

19 To determine the nitrogen content in the crop residues of rye and vetch, the standard values from IPCC (2019a) are used. Based on the description provided by Moore et al. (2014) and Bollero and Bullock (1994), the crop residues are assumed to depict the entire above-ground biomass (see chapter 3.5.1.1). The coefficients are shown in Table A.4 and indicate the crop residues from vetch contain, in total, 40 kg N/ha and of rye, 20 kg N/ha. These values are added to the nitrogen available from the crop residues from corn and soybeans, respectively.

A limitation of IPCC's approach is that it does not distinguish between types of cover crops or their biomass. It only indicates a general coefficient for a system with cover crops and the same system without them. The difference in the soil organic carbon between these systems represents the carbon sequestration potential of this strategy. This change is assumed to occur linearly over 20 years, as described in chapter 3.5.6. Once the 20-year period is over, the soil stops binding additional carbon and its potential as a GHG mitigation strategy is assumed to be completed. The carbon remains bound to the soil only if the grower maintains the practice of cover cropping. If the practice no longer is applied, the carbon is released as GHG. The resulting carbon stocks and sequestration potentials (presented as "annual GHG mitigation potential") are shown in Table 4.13 and the coefficients in Table A.14 in the Appendix.

Table 4.13 Soil organic carbon stocks and sequestration potential with cover crops over 20 years for Typical Farm in Iowa

Case	Soil organic carbon without cover crops ($t\ CO_{2eq}/ha$)	Soil organic carbon with cover crops ($t\ CO_{2eq}/ha$)	Difference ($t\ CO_{2eq}/ha$)	Annual GHG mitigation potential* ($kg\ CO_{2eq}/ha$)
Standard	216	240	24	1,190
Best	216	264	48	2,393
Worst	216	216	0	0

Note: To transform CO_{2eq} to C, the CO_{2eq} value must be divided by 3.67.
Source: own estimation based on IPCC (2019a).

As previously presented in Table 4.12, the nitrogen fixation potential of vetch is estimated as a range that is used for the standard, best and worst cases. Those cases are matched with the corresponding case of the carbon sequestration potential shown in Table 4.13. To exemplify, the highest fixation potential is matched with the highest carbon sequestration potential to determine the best case.

Arguably, the need to produce the seeds for cover cropping results in additional GHG emissions, as these crops are not usually grown in Iowa. These emissions are released from the cultivation of the crop to obtain the seeds to be used in the Typical Farm. Experts' knowledge is used to determine the average yields and corresponding fertilizer and diesel requirements to produce these seeds[20]. The values are used to calculate the GHG emissions, which are then divided per kilogram of seed produced. These emissions are multiplied by the respective seed density of the cover crop used in the Typical Farm. The inputs and the resulting GHG emissions entailed in the production of seeds for the cover crops are shown in Table 4.14.

The estimation of the emissions from rye and vetch is conducted following IPCC (2019a) and coefficients and emission factors are presented in Table A.4 in the Appendix. To determine the adjustment factor for the production of the seeds for cover cropping (see chapter 3.5.1.3), the values from IFA (2020) for the nitrogen content of the harvested parts are used. Rye grains are assumed to have 1.8% nitrogen and vetch, 2.1%. These values are expressed including the moisture of the grain; hence, it is not necessary to calculate the dry matter. Considering the yields, the total removal of nitrogen via harvest is 117 kg N/ha for cereal rye and 53 kg N/ha for hairy vetch. The nitrogen fixation of hairy vetch based on the yield obtained is obtained from experts' knowledge[21] and is assumed to be 85 kg N/ha. This factor is utilized as the only

[20] For rye yield and input use, data are based on information provided by Ronald Haugen from the North Dakota State University (personal communication, 31.05.2021) and from the dataset from *agri benchmark* agri benchmark Cash Crop (2022). The data for vetch is obtained from agricultural analysts of the Spanish National Network of Typical Farms (personal communication, 30.05.2021)

[21] Information provided by Dr. Herwart Boehm from the Thuenen Institute of Organic Farming (personal communication, 02.06.2021).

source of nitrogen relevant to the system, which is used in the adjustment factor. In cereal rye, the only source is the urea used to provide nitrogen.

Table 4.14 Emissions from production of seeds for cover crops for Typical Farm in Iowa

Crop	Yield (t/ha)	Nitrogen (kg N/ha)	Phosphorus (kg P_2O_5/ha)	Potassium (kg K_2O/ha)	Diesel (l/ha)	Emissions (kg CO_{2eq}/t crop)
Rye	6.5	150	30	50	35	**331**
Vetch	2.5	0	40	60	25	**260**

Note: The emission factor used in the Typical Farm is assumed in this calculation. The fertilizers used for the emissions from manufacture are urea for nitrogen, triple superphosphate for phosphorus and potassium chloride for potassium. The GHG emission factors for North America provided by Brentrup et al. (2018) are chosen. For the calculation of vetch, it is assumed that only the seeds are harvested to be sold; the crop residues are incorporated into the soil.
Source: own estimation.

The resulting GHG emissions from cereal rye and hairy vetch seed production are calculated applying the coefficients by IPCC (2019a) and are shown in Table A.4 in the Appendix. The resulting emissions from the production of seeds for cover cropping are 29 kg CO_{2eq}/ha for cereal rye and 9 kg CO_{2eq}/ha. These GHG emissions are added to an emission category named “seeds” under the respective crops. The same values are used in the three cases.

Since this strategy implies changes in corn and soybeans, the estimation of costs and GHG emissions are presented for each crop. Furthermore, given that carbon sequestration occurs only for a limited amount of time, the analysis of this strategy is divided into a transition and a stable stage. The transition stage depicts the 20 years during which the soil sequesters carbon and the stable stage, the situation after the soil has reached a new balance of soil organic content.

Total costs in transition stage from implementing the mitigation strategy

The changes in the cost of using winter cover crops in the Typical Farm during the transition stage of 20 years are presented in Table 4.15. These changes consider the baseline as the reference point. Since the use of cover crops generates different costs in each crop, the changes are presented for each crop individually.

The costs of the seed for the cover crop hairy vetch are represented in the seed costs of corn and the costs for rye in the category soybean. Given that the cover crops are not a part of the baseline, implementing them would increase the expenditure in seeds. The average increase would be 20 USD/ha.

“Fertilizer” in corn is the only category whose change is different across the cases. This results from the assumption of the amount of nitrogen fixed by hairy vetch, which generates different values of reduction in the usage of nitrogen fertilizer. This reduction in corn ranges from 22 to 31 USD/ha. No changes in the fertilizer usage in soybeans are assumed, hence the difference is zero.

The additional pass to terminate the cover crop generates increased costs for herbicide, which are included in the category “pesticides.” The same rate is used in soybean and corn. The additional pass also increases the costs of repairs and maintenance of the machinery as well as diesel and labor in the same magnitude for both crops. The use of contractor services to seed the cover crops also is equal across both crops and all cases (42 USD/ha). The category “liming” is included in “remaining,” as it is not affected by the mitigation strategy.

During the transition stage, this mitigation strategy would increase corn costs by 54 USD/ha in the standard cases, 49 USD/ha in the best case and 58 USD/ha in the worst case. In soybeans, the change is equal across the cases and implies an increase of 76 USD/ha. To estimate the average cost of using cover crops in the

Typical Farm, the simple average of the difference in the costs of corn and soybean is used. As a result, this practice is estimated to increase the costs by an average of 65 USD/ha in the standard case, 63 USD/ha in the best case and 67 USD/ha in the worst case. These are the costs used in the estimation of the mitigation cost per ton of CO_2 reduced.

Table 4.15 Difference in costs using cover crops vs. baseline scenario for Typical Farm in Iowa during transition stage (USD/ha)

	Standard case			Best case			Worst case		
	Corn	**Soybean**	**Avg.**	**Corn**	**Soybean**	**Avg.**	**Corn**	**Soybean**	**Avg.**
Seed	22	18	20	22	18	20	22	18	20
Fertilizer	-26	0	-13	-31	0	-15	-22	0	-11
Pesticides	12	12	12	12	12	12	12	12	12
Diesel	1	1	1	1	1	1	1	1	1
Labor	1	1	1	1	1	1	1	1	1
Machinery	2	2	2	2	2	2	2	2	2
Contractor	42	42	42	42	42	42	42	42	42
Remaining	0	0	0	0	0	0	0	0	0
Total	**54**	**76**	**65**	**49**	**76**	**63**	**58**	**76**	**67**

Source: own estimation.

Emissions per hectare in transition stage from implementing the mitigation strategy

Table 4.16 presents the change in GHG emissions caused by transitioning the Typical Farm from reduced tillage to no-till using the emissions from the baseline as reference. Each crop is presented individually as the changes are different. The emissions from the category “direct” for corn are reduced in all cases. Two of the factors considered in this category are the emissions from crop residues, including the residues from hairy vetch, as well as the emissions triggered by the use of nitrogen fertilizer. The additional residues from the cover crop increase the GHG emissions from this category, as bacteria use the crop residues in their metabolism and release N_2O as a result. However, the reduced input of fertilizer lowers the GHG from this source. The change in emissions from the three cases varies depending on vetch’s assumed nitrogen fixation rate. The best case assumes the highest nitrogen fixation potential, which implies the greatest reduction in the use of nitrogen fertilizer.

In soybeans, the emissions from “direct” are increased in all cases. This results from the crop residues from rye releasing GHG emissions. Contrary to corn, in soybeans there is no source of GHG reduction in the emissions from this category. Hence, the additional emissions from the rye crop residues are not compensated. The emissions from this category are the same in the three cases for soybeans. Overall, for both crops, the change in GHG from “direct” on the average hectare amounts to a net reduction. In the standard case, the decrease totals 42 kg CO_{2eq}/ha; in the best one, 63 kg CO_{2eq}/ha and in the best worst case, 21 kg CO_{2eq}/ha.

For the case of the volatilization and leaching depicted by the “indirect” category, the change in GHG emissions is negative, yet negligible. This applies to all cases and both crops. For corn, as in the case of direct, the lower input of nitrogen fertilizer compensates for the emissions triggered by vetch’s crop residues. In soybeans, the slight reduction in GHG emissions is derived from the added crop residues. They increase the nitrogen the systems loses as direct emissions, which implies that there is less nitrogen available to be volatilized and leached (see chapter 3.5.1.3). The difference in the magnitude between direct and indirect in soybeans is due to the specific emission factors of each category. For the three cases for indirect, the average hectare has a negligible net change.

The emissions from the production of the fertilizer, entailed in the category "manufacture," are reduced in corn due to the reduction of the rate of anhydrous ammonia. The changes differ between the cases based on the reduction in the fertilization rate achieved by the use of the legume cover crop to fixate nitrogen. These range from -85 to -119 kg CO_{2eq}/ha. No change occurs in this source of GHG emissions in soybean. Diesel increases equally in all cases and crops from the seeding of the cover crop and its termination with herbicide.

Land use entails the carbon sequestration promoted by the biomass of the cover crop being incorporated into the soil. The change estimated here is the same for both crops, as the IPCC methodology used to determine the carbon sequestration potential does not distinguish between types of cover crops or the amount of biomass they produce (see Table 4.13). In the standard case, the carbon sequestration shown in land use reduces GHG emissions by 1,189 kg CO_{2eq}/ha, which applies to both crops. In the best case, the sequestration potential is higher than in the standard case, resulting in a reduction in emission of 2,389 kg CO_{2eq}/ha. In the worst case, the IPCC methodology assumes that the soil organic carbon does not change with the inclusion of cover crops, which implies that no reduction in the GHG from the carbon sequestration takes place. However, as previously explained, this scenario is considered as highly unlikely based on Poeplau and Don (2015).

The category "seeds" depicts the emissions from the cultivation to produce the seeds to be used in the cover crops. These GHG emissions are the same across all cases. For corn, the emissions released to produce the seeds of hairy vetch are 9 kg CO_{2eq}/ha. In soybeans with cereal rye as cover crop, the emissions are 29 kg CO_{2eq}/ha.

The average from both crops is used to assess the impact of cover crops in the Typical Farm during the transition stage. In the standard case, GHG emissions are lowered by 1,252 kg CO_{2eq}/ha and in the best case by 2,481 kg CO_{2eq}/ha. In the worst case, given that no carbon sequestration is assumed, but more GHG emissions are released from the crop residues, the emissions from the system with cover crops would be reduced by only 33 kg CO_{2eq}/ha.

Table 4.16 Difference in GHG emissions using cover crops vs. baseline scenario for Typical Farm in Iowa in transition stage (kg CO_{2eq}/ha)

	Standard case			Best case			Worst case		
	Corn	**Soybean**	**Avg.**	**Corn**	**Soybean**	**Avg.**	**Corn**	**Soybean**	**Avg.**
Direct	-141	56	-42	-183	56	-63	-99	56	-21
Indirect	0	-1	0	0	-1	0	0	-1	0
Manufac.	-102	0	-51	-119	0	-60	-85	0	-43
Diesel	12	12	12	12	12	12	12	12	12
Land use	-1,189	-1,189	-1,189	-2,389	-2,389	-2,389	0	0	0
Seeds	9	29	19	9	29	19	9	29	19
Remaining	0	0	0	0	0	0	0	0	0
Total	**-1,411**	**-1,092**	**-1,252**	**-2,670**	**-2,292**	**-2,481**	**-163**	**97**	**-33**

Source: own estimation.

Mitigation costs in transition stage

As a result of implementing hairy vetch and rye as winter cover crops, during the 20 years considered in the transition stage, the cost to mitigate GHG emissions would be 52 USD/t CO_{2eq} in the standard case and 25 USD/t CO_{2eq} in the best case. Despite the unlikeliness of the worst case occurring, it would still offer a mitigation potential of 2,025 USD/t CO_{2eq}. The considerations in the decision to adopt this strategy are discussed at the end of this chapter.

Total costs in stable stage from implementing the mitigation strategy

To maintain the carbon bound in the soil, the use of cover crops must be maintained indefinitely for soybeans and corn. Consequently, due to the continued use of cover crops, the Typical Farm has higher costs than in the baseline scenario. The practices used in the period after the 20-year transition stage are the same in both crops and across all cases. Therefore, there are no differences in the costs of the transition stage and the stable stage. Consequently, the same explanations and implications provided for the transition stage (see Table 4.15) apply here.

As a result of using cover crops in the Typical Farm, the average increase in costs per hectare would range from 65 USD/ha in the best case to 63 USD/ha in the worst case. The standard case implies additional costs of 65 USD/ha.

Emissions per hectare in stable stage from implementing the mitigation strategy

No changes are assumed in the practices in the stable stage compared with the transition stage. Therefore, the changes (see Table 4.16) and their explanations for the individual categories remain unchanged, except for land use. This category includes the changes to the soil organic carbon. According to the methodology used for the estimation of the soil organic carbon, the changes in the carbon stocks occur over 20 years. Therefore, after this period, these values become zero and no more additional carbon is sequestered. This applies to soybeans and corn in the three cases. Consequently, in the stable stage, GHG mitigation is provided only by the nitrogen fixation, compared with the transition stage. The nitrogen fixation potential of hairy vetch is assumed to stay unchanged.

The resulting emissions from corn in the stable stage, for all cases, are negative, as the reduced rate of anhydrous ammonia is enough to compensate for the additional biomass of the cover crop increasing the emissions from the soil. The emissions from manufacture also are reduced compared with the baseline scenario. For corn, the total emissions are -222 kg CO_{2eq}/ha in the standard case, -280 kg CO_{2eq}/ha in the best case and -163 kg CO_{2eq}/ha in the worst case. In soybeans, for all cases, the emissions become positive (97 kg CO_{2eq}/ha), as there is no further carbon sequestration taking place.

As a result, the average hectare of the Typical Farm during the stable stage would continue to have lower emissions. In the standard case, the emission in the stable stage would be 62 kg CO_{2eq}/ha lower than the baseline scenario. The reduction in the best and worst cases are 92 and 33 kg CO_{2eq}/ha, respectively.

Mitigation costs in stable stage and considerations on the adoption of the strategy

Overall, the use of hairy vetch and rye as cover crops in the Typical Farm in the stable stage implies a GHG mitigation cost of 1,039 USD/t CO_{2eq} in the standard case. In the best case, the mitigation cost is 683 USD/t CO_{2eq} and in the worst case, 2,025 USD/t CO_{2eq}. The magnitude of these values is a result of the low reduction of GHG emissions in the stable stage paired with the relatively high costs involved in growing cover crops. Nevertheless, the additional cost during the stable stage can be perceived as the cost of maintaining the carbon bound in the soil because, otherwise, the carbon stored in the soil would be released to the atmosphere as CO_2.

The practice of cover cropping can have additional benefits that are not considered in this assessment. In the short-term, they act as a barrier against wind and rainfall, reducing erosion. Their roots uptake nutrients other than nitrogen, which are released when the crop decomposes, reducing the leaching rates in the winter. In the long-term, the added soil organic carbon content can result in increases in the soil's fertility, which potentially could increase yields, improving the economic return of the cash crops. The increased input of biomass also can improve the soil's structure, increasing its water-holding capacity and likely the yields of the main crop.

Currently, there are cost-share programs available for growers in Iowa to begin or continue using cover crops. For the implementation of this strategy, the existing legal and organizational mechanisms used in these programs could be employed. This has the benefit that the overhead costs of funding the strategy would be lower compared with establishing a new system to distribute the payments to growers. In this regard, assuming this strategy is funded, maintaining the cost-share program is key for the mitigation potential because if the grower stops using the strategy, the previous carbon sequestration would be nullified. A stepwise implementation of the strategy also is feasible, as with the case of no-till (see chapter 4.5.3), which can help support the adoption of this strategy.

Monitoring of this mitigation strategy could be achieved via remote sensing. Satellite imagery could be used to detect the biomass of the cover crop since it grows in a time when the field would have no growing crops without the strategy. Theoretically, the use of this technology would make it possible to determine the type of cover growing as well as its development stage (Hively et al., 2009). This would make the cost of monitoring and enforcing this mitigation strategy comparatively low, as no labor-intensive mechanisms such as field visits are needed.

Comparison of results from Typical Farm with literature

The simulations conducted by McNunn et al. (2020) for the Corn Belt already presented in the comparison in chapter 4.5.2 include the use of cover crops in corn and soybeans. As in the Typical Farm, they evaluated rye as an alternative, which is indicated to reduce emissions by 780 kg CO_{2eq}/ha. In the Typical Farm, this cover crop is used before soybeans and is estimated to mitigate 1,092 kg CO_{2eq}/ha in the standard case of the transition stage. This is the closest of the three cases in terms of mitigation potential.

The same study also includes clover as a legume cover crop that is indicated to abate 393 kg CO_{2eq}/ha. In the Typical Farm, the legume chosen is vetch that is assumed to be used before corn. From three cases evaluated in this Typical Farm, the only case that could be considered similar is the worst one in the transition stage, which mitigates 163 kg CO_{2eq}/ha. As a comparison, in the standard case, the mitigation potential is 1,411 kg CO_{2eq}/ha.

Arguably, the differences can be attributed primarily to the different baseline assumed by the authors. In their study, they consider that the farm initially uses a lower tillage intensity compared with the Typical Farm. Hence, the soil has a lower potential to sequester carbon because it already has a higher carbon content in the baseline.

4.5.5 Enhanced-efficiency fertilizer - nitrification inhibitors (US-4)

The rate at which nitrogen fertilizer transforms in the soil and becomes available to the plant depends on the fertilizer type, weather and soil conditions. This transformation is effected by the soil bacteria and implies that the nitrogen fertilizer becomes more mobile in the soil. The mobility facilitates the crop's uptake of the nutrient, but it also makes it susceptible to losses through leaching, volatilization and emissions. The transformation rate can be slowed down with the use of chemical compounds. The specific chemical depends on the type of nitrogen fertilizer (see chapter 3.6.2.4). In the case of the Typical Farm, the only source of nitrogen for corn is anhydrous ammonia, the transformation rate of which can be reduced with the use of the additive nitrapyrin[22]. Given that slowing down the transformation rate gives the crop more time to take up the nutrient and accumulate it in its structure, less nitrogen is transformed into N_2O or is lost as leaching, reducing GHG emissions. Consequently, given the reduction in the losses, the nitrogen rate can be reduced.

Two additional nitrification inhibitors can be applied in combination with anhydrous ammonia: dicyandiamide and pronitridine. Based on Akiyama et al. (2010), dicyandiamide's efficiency at reducing N_2O emissions is lower than that of nitrapyrin. Literature studying the effect of pronitridine on N_2O emissions from anhydrous ammonia is scarce as the product was approved in 2018 for use in the American market. No applicable results testing its effect as a GHG mitigation source were identified. Consequently, the analysis of this mitigation strategy is conducted using nitrapyrin.

Nitrapyrin is generally recommended in Iowa, especially for fall application of manure or synthetic fertilizer, as it reduces the losses of nitrogen that occur in winter and early spring before the crop is seeded (Parkin and Hatfield, 2010). Applying nitrapyrin does not require any adjustment or changes to the application process or schedule, and it only needs to be mixed with the anhydrous ammonia in the tank. However, the focus group stated they do not use nitrapyrin as they consider it an expensive input, especially if it is for a spring application of nitrogen, as losses are estimated to be lower than from a fall application. The use implies an additional cost of 22 USD/ha. Considering a price of nitrogen of 1 USD/kg, for the price of the application of nitrapyrin, 28 kg N/ha could be added instead.

Akiyama et al. (2010) conducted a meta-analysis to assess the effect of various nitrification inhibitors based on the type of inhibitors and the source of nitrogen. They concluded that nitrapyrin reduced emissions of N_2O by 31% to 44%. Wolt (2004) conducted a similar meta-analysis and determined the effect to be 51%. However, both studies analyzed the effect across multiple crops, fertilizer rates and application timings. Considering that the share of N_2O emissions reduced is larger in fall applications due to the absence of the crop, combining the results from fall and spring fertilizer applications may generate misleading results. Furthermore, yearly differences in precipitation and temperatures also are indicated to determine the effect of nitrapyrin in N_2O emissions (Bremner et al., 1981). Hence, the inclusion of short-term studies in these meta-analyses also increases the variation in the results. No long-term studies assessing the effect on GHG emissions of nitrapyrin applied with anhydrous ammonia in conditions similar to the Typical Farm could be identified[23]. Consequently, it is assumed that the results from the meta-analyses must be adapted to be applicable.

The values from the meta-analysis by Wolt (2004) are used as reference to determine the change in nitrogen losses as N_2O. The studies included in this meta-analysis emphasized the Corn Belt region. In comparison, Akiyama et al. (2010) includes studies from multiple climates that do not represent Iowa or neighboring states. This is considered relevant due to the effect of climate on the efficiency of nitrapyrin to reduce emissions (Akiyama et al., 2010). Therefore, based on Wolt (2004), the reduction of nitrogen losses as N_2O is assumed to be 51%. The standard error of the analysis determines the range of values for the best and worst cases. This results in a maximum reduction of 55% and a minimum reduction of 47%. To adapt these values to the Typical Farm, it is assumed that they reduce the nitrogen losses as N_2O. These emissions are depicted by the category "direct" - the GHG emissions from the nitrogen cycle in the soil (see chapter 3.5.1.1). It is assumed that reducing the losses implies a need to apply less nitrogen as fertilizer since a smaller share of the nutrient applied is lost to the environment. Therefore, the reduction in the nitrogen

[22] Nitrapyrin is the common name of the chemical compound used.

[23] The study by Parkin and Hatfield (2010) measuring emissions from anhydrous ammonia with nitrapyrin in corn in Iowa is not considered. This study is deemed as short-term. Furthermore, the results in this study do not produce statistically significant results for the effect of nitrapyrin in GHG emissions from spring applications of nitrogen.

rate is obtained by multiplying the nitrogen loss from direct in the baseline scenario (US-0) with the estimated reduction in emissions from nitrapyrin.

Furthermore, Yadav (1997) measured the leaching of nitrogen from corn cultivation in three different locations for a period of four years using multiple sources of nitrogen at varying rates. Among others, the effect of nitrapyrin combined with anhydrous ammonia applied in spring was tested in two plots. Based on the results, a statistical regression was calculated to account for inter-year variability. The test sites have soil and climate conditions similar to the Typical Farm[24]. The effect of nitrapyrin was statistically significant for only one of the plots and indicates an average reduction in leaching of 25%. The author states that no negative effects on corn yields were reported. No further applicable results could be identified in the literature. Consequently, it is assumed that the use of nitrapyrin in the Typical Farm also reduces the nitrogen losses via leaching by 25%. The author does not provide a range that can be used for the best and worst cases; hence the same value is assumed for the three cases. These losses are depicted in the category “indirect,” along with the losses from volatilization (see chapter 3.5.1.3). As with the reduction applied to the nitrogen losses from direct emissions, the 25% reduction in leaching is assumed to be the potential reduction to reduce the nitrogen rate. This potential reduction is the result of multiplying the losses from leaching in the baseline by the reduction in leaching from nitrapyrin.

For the calculation of the total fertilizer reduction potential attainable by using nitrapyrin, the effect on emissions in “direct” and on leaching are added. Since nitrapyrin reduces the losses of nitrogen from the soil, the total reduction in the fertilization rate is assumed to not cause changes in corn yields because the availability of nitrogen in the soil is not affected. The values and reduction of each case are presented in Table 4.17. The total reduction shown in the last column is applied to the fertilizer rate on the Typical Farm. This value also is used to reduce the adjustment factor applied to indirect emissions (see chapter 3.5.1.2).

Table 4.17 Effect of nitrapyrin and nitrogen fertilizer reduction for the Typical Farm in Iowa

Case	Emissions from direct			Leaching			Total fertilizer reduction
	N loss	Share reduced	Fertilizer reduction	N loss	Share reduced	Fertilizer reduction	
	(kg N/ha)	(%)	(kg N/ha)	(kg N/ha)	(%)	(kg N/ha)	(kg N/ha)
Standard	3.5	51	1.8	30.5	25	7.6	**9**
Best	3.5	55	1.9	30.5	25	7.6	**10**
Worst	3.5	47	1.6	30.5	25	7.6	**9**

Source: own estimation.

An alternative to reducing the fertilizer rate would be to maintain the nitrogen rate unchanged when applying nitrapyrin, which theoretically could increase yields, as reported in some studies (Akiyama et al., 2010; Wolt, 2000, 2004). This occurs because reducing nitrogen losses results in additional nitrogen becoming available to the crop, despite the same rate being applied. However, the conditions under which this happens, and the magnitude of the change are not yet completely understood, especially since nitrapyrin has been shown to promote the accumulation of nitrogen in the soil (Wolt, 2004). Therefore, it is not possible to determine the effect that the perceived additional nitrogen available to the crop would have on corn yields.

[24] The field where the testing took place is in the south of Minnesota, the state located directly to the north of Iowa.

Total costs from implementing the mitigation strategy

All the costs are the same as in the baseline scenario (US-0) shown in Table 4.3 with exception of the cost of fertilizer. The use of nitrapyrin lowers the nitrogen rate needed to maintain the corn yield, which reduces costs of this category (-7 USD/ha). Yet, the cost of the nitrification inhibitor (22 USD/ha), included in the same category, is higher than the savings from lowering the input of anhydrous ammonia. The resulting change is an increase of 15 USD/ha in the cost of fertilizers. The cost is approximately the same in all cases, as the only difference between them is less than 1 kg of fertilizer. Given that no other category has changes, these results depict the cost of implementation of this GHG mitigation strategy in the Typical Farm.

Therefore, this GHG mitigation strategy generates additional costs; the cost of utilizing the nitrification inhibitor in corn implies a change of 15 USD/ha in the three cases. The exact differences between the cases are not shown due to rounding, but the difference between the best and worst case is less than 0.5 USD/ha.

Emissions per hectare from implementing the mitigation strategy

Table 4.18 presents the resulting GHG emissions per hectare from producing corn with the nitrification inhibitor on the Typical Farm and the comparison with the baseline scenario. The nitrification inhibitor lowers the emissions in the category “direct” as a result of lowering the nitrogen fertilizer input. The reduction is 70 kg CO_{2eq}/ha in the standard case with minuscule differences in the best and worst cases.

The emissions entailed in the category “indirect” experience the greatest reduction in GHG emissions. As with direct emissions, this is the result of lowering the fertilizer rate, which implies that less nitrogen is available in the system. The reduction in emissions from this category is 50 kg CO_{2eq}/ha in the standard case. The best and worst cases have a difference of 1 kg CO_{2eq}/ha compared with the standard case. The emissions from the manufacture of fertilizer are reduced as well. This reduction results from the lowered use of anhydrous ammonia. This reduction totals 29 kg CO_{2eq}/ha in the three cases.

The category “remaining” includes liming, urea, diesel and land use. “Diesel” is the only category from this list that has emissions different than zero.

As a result of using the nitrification inhibitor in the Typical Farm, the GHG emissions are estimated to be reduced by 149 kg CO_{2eq}/ha in the standard case, by 148 kg CO_{2eq}/ha in the worst case and by 152 kg CO_{2eq}/ha in the best case. These are the values used to calculate the mitigation cost per ton of CO_{2eq}.

Table 4.18 GHG emissions in corn using nitrification inhibitors for the Typical Farm in Iowa (kg CO_{2eq}/ha)

	Baseline scenario	Standard case		Best case		Worst case	
		Strategy	Change	Strategy	Change	Strategy	Change
Direct	1,647	1,577	-70	1,575	-72	1,577	-70
Indirect	200	150	-50	148	-51	150	-49
Manufacture	606	578	-29	578	-29	578	-29
Remaining	210	210	0	210	0	210	0
Total	**2,663**	**2,514**	**-149**	**2,511**	**-152**	**2,515**	**-148**

Source: own estimation.

Mitigation costs and considerations on the adoption, monitoring and enforcement of the strategy

The use of nitrification inhibitors in the production of corn on the Typical Farm in Iowa results in a GHG mitigation cost of 100 USD/t CO_{2eq} in the standard case. The best-case scenario implies a mitigation cost of 98 USD/t CO_{2eq} and the worst case, 101 USD/t CO_{2eq}.

Using nitrification inhibitors does not imply investments in technology or changes in growers' operations or production strategies. Hence, it can be assumed that the implementation of this GHG mitigation strategy could be achieved in a short time frame. Since there is no learning process for growers, support via extension services would not necessarily need to be expanded as a result of the strategy. As with the optimization of the nitrogen rate (see chapter 4.5.2), this strategy could be adopted on a fraction of the acreage and increased gradually stepwise.

An additional benefit that is not directly covered in this assessment is reduction of NO_3 in water caused by leaching. The methodology by IPCC (2019a) considers leaching due to N_2O emissions that are released from water bodies containing nitrates derived from fertilization. However, nitrates in the water pose an environmental risk as they can cause eutrophication. In some regions in Iowa, the concentration of nitrates in water has reached levels harmful for humans (Jones et al., 2016). Hence, the installation of filtering equipment to make the water potable and safe has been necessary. Since nitrification inhibitors reduce leaching, they lower the concentration of nitrates in water. This reduces the environmental impact of nitrogen fertilization beyond the reduction of GHG. Hence, this strategy would have a positive externality.

Monitoring the use of nitrification inhibitors in the field is possible. However, this requires taking soil samples to measure the concentration of nitrapyrin. A limitation is that nitrapyrin could degrade at different rates in the soil depending on weather (Wolt, 2000). Measuring its application therefore would be challenging, as it may degrade before the soil sampling is conducted. This could pose a limitation if a random soil sampling is part of the monitoring used by the controlling authority. Alternatively, it could be enforced that all anhydrous ammonia is sold already mixed with nitrapyrin. This would have the advantage that monitoring at farm-level would not be necessary. Furthermore, assuming a subsidy is used to cover the additional costs, the fertilizer could be sold at the price that it would have without the nitrification inhibitor.

Introducing a tax on synthetic nitrogen would increase the price of the fertilizer, which could promote the adoption of nitrification inhibitors. Since nitrogen would become more expensive, the cost entailed in the loss of nitrogen as leaching would be higher. Thus, using inhibitors would imply a greater reduction in the expenditure of fertilizer, which would make them comparatively more economical.

Comparison of results from Typical Farm with literature

The report by ICF (2013), already presented in the comparison in chapter 4.5.2, calculated the mitigation potential of nitrification inhibitors to be 120 kg CO_{2eq}/ha. This abatement is similar but lower than the worst case assumed for corn in the Typical Farm, which is estimated to be 148 kg CO_{2eq}/ha. However, the mitigation costs from the report are 63 USD/t CO_{2eq}, which is lower than even the best case of the Typical Farm (98 USD/t CO_{2eq}). As previously presented, this is derived from the assumptions regarding the cost of the input, especially the inhibitor, as well as the mitigation potential considered.

4.6 Combination of strategies (US-C)

Given the urgency to mitigate emissions to abate the effects of climate change, it is key to determine the highest emissions savings potential achievable in a short time. This is achieved by combining multiple mitigation strategies on the same acreage at the same time to stack their mitigation potentials. Yet, the combinations may generate repercussions that must be addressed, both on the operational and economic levels as well as in terms of GHG emissions. Thus, the objective of this section is to assess the barriers and synergies that may arise when implementing a combination of strategies to determine feasible options. The evaluation of the combinations assumed as feasible and their resulting costs and GHG mitigation are assessed in the next section.

To assess the limitations and synergies that may occur in the combination, the key changes and implications of each strategy evaluated are presented and discussed. The changes are compared with the baseline scenario (US-0) of the Typical Farm in Iowa.

Arguably, the use of nitrification inhibitors (US-4) is the strategy that has the lowest impact regarding technical changes or effects on the soil, the crop, or in the rotation because it does not imply changes in the scheduling or realization of the operations. Thus, given the relative simplicity of its implementation, the usage of the inhibitor would not interfere with any of the farm operations entailed in the other strategies. Thus, it is theoretically feasible to combine this strategy with any other. Therefore, concerning the combination, it is deemed that this mitigation strategy does not require further assessment.

Regarding the mechanization and scheduling of farm operations, optimizing nitrogen (US-1) fertilization involves an additional pass to spread fertilizer, which is assumed to occur after the crop has been seeded. Using cover crops (US-3) entails two additional farm operations: seeding the crop and terminating it. Seeding the crop is done by a contractor and occurs after harvest of the cash crop. Termination takes place before spring tillage operations, which are done to prepare the soil for seeding. Thus, no competition for labor or machinery derives from implementing these two strategies together. Transitioning to no-till (US-2) implies stopping the tillage of the soil. These tillage operations entailed in the baseline scenario are not directly required for the accomplishment of the other mitigation strategies. Consequently, with regard to mechanization, it is feasible to combine all the strategies as no negative interaction could be identified that would impact the performance of the mitigation strategies.

No relevant effects on the soil or the crop management, which could interact with other strategies, are assumed from the additional fertilizer pass to optimize the efficiency of nitrogen. However, combining no-till with cover cropping implies that more biomass is left on the soil's surface than in the individual strategies. These combined residues would take longer to decompose and block a larger share of the sunlight from reaching the soil. Consequently, the conditions for the seeds of the summer crop to germinate would worsen, possibly lowering yields. Dozier et al. (2017) tested the effect of multiple species of cover crops, including rye and hairy vetch, in corn and soybean yields across different tillage intensities. The tests were conducted over three years and with four replications. No statistically significant effect on yield was identified for the combination of cover crops with no-till. Moreover, they report that tillage did not affect the biomass of the cover crops. Thus, the nitrogen supply provided by hairy vetch toward corn would not be reduced. No further applicable scientific literature assessing the combined effect of cover crops and tillage in conditions similar to this Typical Farm could be identified. Consequently, following Dozier et al. (2017), it is assumed that no relevant interaction that alters yields occurs from the combination of these two strategies.

No further limiting interactions are identified in this assessment. Consequently, it is assumed that it is feasible to implement the combination of all the strategies without negatively affecting yields or other performance indicators on the Typical Farm in Iowa.

The resulting changes in GHG emissions and associated costs are presented in the following section. Only the elements for which the combination leads to interactions causing effects different from the individual strategy are discussed. Thus, the elements of the individual strategies not mentioned are assumed to remain unaffected by the combination.

As with the previous analysis, the combination is compared against the baseline scenario. The estimation of the standard, best and worst cases for the combinations is conducted by matching the corresponding case of each individual strategy. For instance, the best case for no-till is paired with the best case for the optimization of nitrogen and from nitrification inhibitors to establish the best case of the combination.

Costs and mitigation potential of optimization of nitrogen rate, no-till, use of cover crops and nitrification inhibitors combined

Combining all the mitigation strategies would generate two sources of GHG reduction: lowering the emissions triggered by nitrogen use and carbon sequestration in the soil. Both sources are affected by more than one strategy. Thus, the interactions in the combinations must be assessed. The carbon sequestration potential attributable to the strategies is realized over 20 years. Thus, the assessment is divided into a transition stage and a stable stage. Moreover, the effects on corn and soybean are presented because some strategies affect both crops.

The strategies affecting emissions from nitrogen are the optimization of the nitrogen rate (US-1), hairy vetch as a cover crop (US-3), and nitrification inhibitors (US-4). These three strategies have different approaches to reducing emissions, which have already been explained in detail in the respective chapters. However, the total change in the nitrogen rate from their combination must be determined.

The strategy of optimization of nitrogen rate focuses on lowering the fertilizer rate to the minimum needed for yields to remain constant and avoid soil mining, which occurs when not enough nitrogen is supplied to cover the requirements of the crop and the losses from the soil. However, the use of nitrification inhibitors reduces the proportion of nitrogen losses, increasing the share of the nitrogen application that remains in the soil and can be used by the crop. Thus, adding nitrification inhibitors to the optimized nitrogen rate would enable lowering it even more (see chapter 4.5.2) since the losses from the system would be reduced without decreasing the availability of the nutrient for the crop.

Moreover, this rate of nitrogen could be further reduced by discounting the nitrogen provided by hairy vetch. The nitrogen mineralized from this cover crop is assumed to replace a share of the requirement of synthetic nitrogen in corn (see chapter 4.5.4). The mineralization rate of the nitrogen in the hairy vetch residues is unaffected by the nitrification inhibitor (Chalk et al., 1990; Crawford and Chalk, 1992).

To calculate the total nitrogen reduction in the combination, the nitrogen from fixation by hairy vetch (US-4) is deducted from the optimized nitrogen rate (US-1) to determine a preliminary fertilizer rate. The GHG emissions from this preliminary rate are added to the emissions triggered by the nitrogen in the crop residues from hairy vetch. This indicates the total losses on which the nitrification inhibitor acts. Following the same procedure and coefficients presented in chapter 4.5.5, nitrapyrin reduction of nitrogen losses as emissions from direct and leaching then is applied. The total nitrogen reduction from nitrapyrin is presented in Table B.4 in the Appendix.

The resulting nitrogen rate used in the combination is shown in Table 4.19. The reduction from the nitrification inhibitor shown in the fourth column is the lowest in the best case. As previously explained, the best case means that the highest reduction in the nitrogen rate is assumed. In turn, this implies that this strategy leaves the least amount of nitrogen susceptible to being lost. Hence, the amount of nitrogen on which nitrapyrin has an effect is the lowest in the best case. Therefore, the nitrogen reduction potential from the nitrification inhibitors in the best case of the combination is the lowest of the three cases, despite assuming the highest reduction of nitrogen losses by the nitrification inhibitor (see third column in Table B.4 in the Appendix). Equivalently, the worst case results in the highest reduction potential from nitrification inhibitors.

Table 4.19 Nitrogen rate in combination of strategies for Typical Farm in Iowa

Case	Rate in Baseline (US-0)	Reduction of nitrogen rate			Rate in Combination (US-C)
		Optimization of nitrogen (US-1)	Cover crop (US-3)	Nitrification Inhibitors (US-4)	
	(kg N/ha)	(kg N/ha)	(kg N/ha)	(kg N/ha)	(kg N/ha)
Standard	181	15	34	6.4	126
Best	181	25	39	4.3	112
Worst	181	5	28	8.5	139

Source: own estimation.

No applicable studies testing the effect on corn yields from these combined strategies, or the resulting rate, could be identified. However, the meta-analysis from Wolt (2004), which was discussed in chapter 3.6.2.4, can be used as a reference. Several results included in this study utilize nitrogen rates considerably lower than what is assumed in this case. Among them are the results from Cerrato and Blackmer (1990) for corn in Iowa. In most of the trials, no change or a positive one is reported by the authors, although it is indicated that the yield was reduced in some cases. Consequently, while there is a degree of uncertainty on the specific conditions in which nitrapyrin may influence corn yield, it is assumed that no interaction with a negative outcome on the yields occurs in this case.

The carbon sequestration potential in the combination can be calculated following the IPCC (2019a) methodology. Their approach already was used to determine the potential of transitioning to no-till (US-2) and cover cropping (US-3) individually. The coefficients used in this case as well as the resulting carbon stock of the system under the combination are shown in Table A.14 in the Appendix. The mitigation potential used in the combination of strategies is shown in Table 4.20.

Table 4.20 Soil organic carbon stocks and sequestration potential with combination of strategies over 20 years for Typical Farm in Iowa

Case	Soil organic carbon without combination	Soil organic carbon with combination	Difference	Annual GHG mitigation potential*
	(t CO_{2eq}/ha)	(t CO_{2eq}/ha)	(t CO_{2eq}/ha)	(kg CO_{2eq}/ha)
Standard	216	252	35	1,768
Best	216	288	72	3,580
Worst	216	218	1	67

Source: own estimation.

According to IPCC (2019a), the carbon sequestration potential of the combination is greater than the potential of the two individual strategies, which can be interpreted as a synergistic effect[25] (see chapter 3.5.6). Calegari et al. (2008) tested multiple cover crops across no-till and tillage systems over 19 years in a corn–soybean rotation in southern Brazil. The results indicated that the implementation of both strategies (no-till and cover cropping) produced the highest gain in soil organic carbon. Although the crop rotation is not mentioned, similar results are reported for Denmark by Abdollahi and Munkholm (2014) for fields receiving a similar treatment for 10 years. No long-term studies testing these factors in climatic conditions

[25] The approach is based on multiplication of coefficients. Tillage management and the input of organic matter, which depicts the use of cover crops, among other elements, have their own coefficient. While in the individual strategies only one of the coefficients is modified, in the combination both are changed. Therefore, they multiply with each other, resulting in a value greater than their sum (see Table A.14 in the Appendix).

similar to Iowa could be identified. Hence, although these two studies resemble only partly the situation of this Typical Farm, they are considered an indication of the plausibility of the synergies generated by the approach used in this analysis.

Total costs in transition stage from implementing the mitigation strategy

The changes in the cost of optimizing the nitrogen rate, using nitrification inhibitors, transitioning to no-till (including the yield penalty) and establishing cover crops on the Typical Farm in Iowa are shown in Table 4.21. The values represent the 20-year transition stage.

Seed costs are increased due to the use of hairy vetch and cereal rye to establish cover crops. This cost on the average hectare is 20 USD/ha and is the same across the standard, best and worst cases. The cost of fertilizer is reduced in corn, which results from lowering the use of anhydrous ammonia. The variation in the reduction in cost observed in the three cases depends on the nitrogen fixation potential of hairy vetch, the reduction from nitrapyrin and the reduction from the optimization of the nitrogen rate. The cost of nitrapyrin is included in this category and is equal in the three cases. No change in this category is observed in soybeans. The total cost reduction from fertilizer for the average hectare is 10 USD/ha in the standard case and ranges from 16 to 5 USD/ha for the best and worst cases.

The additional costs of pesticides are the same in corn and soybeans (12 USD/ha) in all cases. This is derived from the additional pass to terminate the cover crop before seeding the summer crops. The categories diesel, labor and machinery have a decreased cost. Although the combination implies added costs from additional applications of pesticides and fertilizers, transitioning to no-till reduces these costs comparatively more. The sum of these three categories implies a reduction of 23 USD/ha on the average hectare in the three cases. Similarly, for both crops, the expenditure on contractors, which are hired to seed the cover crops, increases by 42 USD/ha. Likewise, the foregone revenue from the yield penalty is the same in all cases, averaging 34 USD/ha. Liming is included in the category “remaining,” which is not affected by the mitigation strategies.

As a result of applying the combination of strategies, the total for the average hectare would be an increase in the cost of 76 USD/ha in the standard case, 70 USD/ha in the best case and 81 USD/ha in the worst case. These costs are used for the estimation of the mitigation cost per ton of CO_2.

Table 4.21 Difference in costs combination vs. baseline scenario for Typical Farm in Iowa in transition stage (USD/ha)

	Standard case			Best case			Worst case		
	Corn	**Soybean**	**Avg.**	**Corn**	**Soybean**	**Avg.**	**Corn**	**Soybean**	**Avg.**
Seed	22	18	20	22	18	20	22	18	20
Fertilizer	-21	0	-10	-31	0	-16	-10	0	-5
Pesticides	12	12	12	12	12	12	12	12	12
Diesel	-8	-3	-6	-8	-3	-6	-8	-3	-6
Labor	-5	-1	-3	-5	-1	-3	-5	-1	-3
Machinery	-20	-8	-14	-20	-8	-14	-20	-8	-14
Contractor	42	42	42	42	42	42	42	42	42
Foregone revenue	55	13	34	55	13	34	55	13	34
Remaining	0	0	0	0	0	0	0	0	0
Total	**79**	**73**	**76**	**68**	**73**	**70**	**89**	**73**	**81**

Source: own estimation.

Emissions per hectare in transition stage from implementing the mitigation strategy

Table 4.22 shows the GHG emissions during the 20-year transition period from combining the GHG mitigation strategies. The emissions from direct and indirect in corn are considerably reduced in all three scenarios. These emissions are lowered by the reduction of nitrogen fertilizer input, which is one of the main drivers of the GHG emissions from these categories. The yield penalty from no-till results in fewer crop residues, yet the added biomass from the cover crops increases the total nitrogen in residues compared with the baseline. However, compared to the reduction from the lowered nitrogen rate, this increase is negligible. The combined effect of direct and indirect in corn is a reduction of 414 kg CO_{2eq}/ha in the standard case, 557 kg CO_{2eq}/ha in the best case, and 272 kg CO_{2eq}/ha in the worst case. Since the best case assumes the largest reduction in nitrogen, it also results in the highest mitigation of emissions from these categories.

The difference in GHG released from these categories in soybeans is the same in the three cases. The change is principally a product of the additional biomass from the cover crop, which results in additional crop residues available for the bacteria to decompose, increasing the release of GHG. The yield penalty implies that less nitrogen is available to be lost, as the fixation potential is determined as a function of the soybean yield (see chapter 4.5.1). This slightly lowers the indirect emissions. Nevertheless, the magnitude of the change can be considered negligible. The total change from direct and indirect in soybeans is 55 kg CO_{2eq}/ha. The category "seeds" depicts the emissions from the cultivation of hairy vetch and rye to obtain the seeds to use as cover crops. These are the same across the three cases and, on average for both crops, are 19 kg CO_{2eq}/ha.

The emissions from manufacture also are reduced in corn, as the intensity of the use of fertilizer is reduced. No changes occur in soybeans. The variation in the values is the result of the different assumed potentials to reduce nitrogen fertilization. The emissions from diesel also are reduced in all cases for both crops. In corn, the added pass to spread fertilizer increases diesel GHG emissions yet ceasing to conduct the tillage operations lowers the emissions comparatively more. In soybeans, only the tillage operations are no longer conducted compared with the baseline scenario. Seeding and termination of the cover crops also increases the emissions from these categories in the same magnitude for both crops. The average hectare has a reduction of 20 kg CO_{2eq}/ha in emissions from this category.

"Land use" is the category that offers the largest reduction in emissions. The total reduction from this category for the average hectare is 1,709 kg CO_{2eq}/ha in the standard case, 3,520 kg CO_{2eq}/ha in the best case, and only 10 kg CO_{2eq}/ha in the worst case. The emissions from ILUC from the yield penalty (57 kg CO_{2eq}/ha in the average hectare) already are included in this value. The low sequestration value in the worst case is partly due to the approach used by IPCC (2019a), which is discussed in chapter 4.5.4. The category "remaining" includes liming and urea, which are not affected by the combination of the GHG mitigation strategies.

In the transition stage of the combination of strategies, the average hectare would have a reduction of 1,973 kg CO_{2eq}/ha, 3,876 kg CO_{2eq}/ha in the best case, and 182 kg CO_{2eq}/ha in the worst case. These values are used to determine the mitigation cost per ton of CO_2.

Table 4.22 Difference in GHG emissions combination of strategies vs. baseline scenario for Typical Farm in Iowa in transition stage (kg CO_{2eq}/ha)

	Standard case			Best case			Worst case		
	Corn	**Soybean**	**Avg.**	**Corn**	**Soybean**	**Avg.**	**Corn**	**Soybean**	**Avg.**
Direct	-301	56	-123	-403	56	-173	-200	56	-72
Indirect	-113	-1	-57	-154	-1	-78	-72	-1	-36
Seeds	9	29	19	9	29	19	9	29	19
Manufac.	-167	0	-83	-208	0	-104	-126	0	-63
Diesel	-32	-8	-20	-32	-8	-20	-32	-8	-20
Land use	-1,665	-1,754	-1,709	-3,476	-3,565	-3,520	34	-54	-10
Remain.	0	0	0	0	0	0	0	0	0
Total	**-2,269**	**-1,677**	**-1,973**	**-4,263**	**-3,488**	**-3,876**	**-386**	**22**	**-182**

Source: own estimation.

Mitigation costs in transition stage

Thereby, with the combination of strategies, the mitigation costs in the transition stage would be 38 USD/t CO_{2eq} in the standard case, 18 USD/t CO_{2eq} in the best case and 446 USD/t CO_{2eq} in the worst case.

Total costs in stable stage from implementing the mitigation strategy

The Typical Farm must continue to use no-till and cover crops after the carbon sequestration potential is fulfilled, or the carbon bound to the soil will be released as GHG. Hence, the same operations are assumed. Therefore, the resulting costs of the combination of mitigation strategies are the same as during the transition stage (see Table 4.21). The sole difference is the disappearance of the yield penalty indicating that foregone revenue becomes zero.

In consequence, the average hectare of the Typical Farm has a net increase in costs of 42 USD/ha in the standard case. The increase ranges from 36 to 47 USD/ha in the best and worst case, respectively. The difference in the values depends only on the assumed fertilizer reduction potential.

Emissions per hectare in stable stage from implementing the mitigation strategy

During the stable stage in the combination of strategies, the soil no longer can bind additional carbon. Moreover, given that the yield penalty no longer applies, no more emissions from ILUC are assumed. Therefore, the emissions entailed in land use become zero. This is the only difference in GHG emissions compared with the transition stage (see Table 4.22).

As a result, the emissions on the average hectare are lowered by 257 kg CO_{2eq} /ha in the standard case, by 349 kg CO_{2eq}/ha in the best case, and by 165 kg CO_{2eq}/ha in the worst case. This implies that after the carbon sequestration potential is fulfilled, the Typical Farm would continue to have lower emissions compared with the baseline scenario. As with the difference in costs, the variation in these results observed across the cases depends on the reduction of nitrogen fertilizer assumed.

Mitigation costs in stable stage and considerations on the adoption of the strategy

During the stable stage in the combination of strategies, the mitigation cost for the Typical Farm is 162 USD/t CO_{2eq} in the standard case. The range of the cost is 104 USD/t CO_{2eq} to 284 USD/t CO_{2eq} in the best and worst cases.

None of the farm operations entailed in the individual mitigation strategies compete for the same machinery or time schedule during the season when applied in combination. The only investment in machinery is corn stalk stompers, the price of which can be considered low. Hence, it can be argued that growers could implement the combination of strategies within a short timeframe on a wide scale.

The combination of strategies does not alter the considerations presented regarding the monitoring and enforcement of the individual mitigation strategies. Therefore, the implementation of the single elements included in the combination would still need to be monitored independently. Likewise, the same approaches to promote the adoption of the individual strategies are valid in the combination. Moreover, to facilitate adoption, the strategies, either individually or in combination, could be implemented in a stepwise process, as discussed in chapter 4.5.2. Consequently, these aspects are not presented here as they already have been addressed in the corresponding chapters.

Nonetheless, the adoption of certain strategies could be promoted with the same approach, such as a nitrogen tax improving the economics of nitrification inhibitors and the optimization of nitrogen fertilization. Similarly, certain strategies could share the monitoring mechanism; e.g., no-till and cover crops could be policed via remote sensing. Arguably, a considerable share of these transaction costs is fixed for the controlling entity. Therefore, promoting and monitoring the combination of strategies theoretically would cost less than the sum of the transaction costs attached to the individual strategies. This could increase the likelihood of policymakers deciding toward implementing the combination over individual strategies, as they could perceive increased GHG reduction with a lower transaction cost per ton of CO_2 abated.

4.7 Policy advice: ranking of GHG mitigation strategies

The summary of the costs to mitigate one ton of CO_2 for all the strategies assessed in the Typical Farm in Iowa are displayed in Table 4.23. Gradually implementing two or more strategies at the same time, as indicated in this ranking, may generate interactions that are not accounted for. These may positively or negatively affect the cost efficiency of the strategies. Moreover, the combination is included in this ranking, which by definition entails all the strategies assessed in the Typical Farm. Thus, combining this strategy with individual strategies is not possible. Nevertheless, including this strategy in the ranking indicates at what point it is more economically efficient to adopt all the strategies instead of individual ones, as the individual strategies would be more expensive and offer a lower GHG mitigation potential.

The results are presented for an average hectare in the standard, best and worst cases. For the strategies that influence one corn production, such as nitrification inhibitors (US-4), the changes in emissions and costs for soybeans are assumed to be zero. A simple average is then calculated. Expressing the values for an average hectare enables the interpretation of the results at the farm level, as opposed to only one crop.

The strategies are divided into short-term and long-term perspectives, as presented in chapter 3.8. The mitigation strategies are ranked based on their mitigation cost-efficiency to provide policy advice regarding which strategies should be studied further to determine likely transaction costs. As presented in chapter 2.6, it is recommended that policymakers utilize the ranking of strategies in the short-term perspective because of the urgency to achieve the goals in the Paris Agreement until 2050.

Table 4.23 Comparison of mitigation costs in the standard, best and worst cases for the average hectare in Typical Farm in Iowa (USD/t CO_{2eq})

Strategy	Code	Standard case	Best case	Worst case
Optimization of nitrogen rate	US-1	-16	-29	74
Reduction of tillage intensity	US-2			
Transition stage (short-term)		8	4	89
Stable stage		-739	-739	-739
100-year average (long-term)		-177	-105	-564
Cover crops	US-3			
Transition stage (short-term)		52	25	2,025
Stable stage		1,039	683	2,025
100-year average (long-term)		216	110	2,025
Enhanced efficiency fertilizer (inhibitors)	US-4	100	98	101
Combination	US-C			
Transition stage (short-term)		38	18	446
Stable stage		162	104	284
100-year average (long-term)		81	41	319

Source: own estimation.

Considering a short-time cost-efficiency (transition stage), the optimization of the nitrogen rate becomes the most economic mitigation alternative (-16 USD/t CO_{2eq}) and is the only strategy with a negative cost. Reduction to no-till has a positive cost, although it can be regarded as low (8 USD/t CO_{2eq}) compared with the other strategies. The combination of strategies has the next highest cost, followed by cover crops, with 38 USD/t CO_{2eq} and 52 USD/t CO_{2eq}, respectively. The enhanced efficiency fertilizer (nitrification inhibitors) is the least efficient strategy in economic terms (100 USD/t CO_{2eq}) in the short term.

Using the annualized 100-year average of the standard case for the long-term perspective, reducing tillage to no-till offers the lowest mitigation cost (-177 USD/t CO_{2eq}). The optimization of the nitrogen rate results in the second-lowest mitigation cost (-16 USD/t CO_{2eq}). These strategies are the only win-win scenarios. The combination of strategies is the third-best alternative (81 USD/t CO_{2eq}) followed by nitrification inhibitors (100 USD/t CO_{2eq}). Finally, the use of cover crops has the highest mitigation cost on average, which is significantly larger than any other strategy (216 USD/t CO_{2eq}).

The ranking of strategies based on mitigation cost does not differ when comparing the standard and best cases in the long- and short-term perspectives. The ranking in the worst case in the short-term perspective is comparable, but nitrification inhibitors are the third most efficient strategy, followed by the combination and lastly, cover crops. In the long term, assuming the worst case, the ranking of cost-efficiency is reduction of tillage intensity, optimization of the nitrogen rate, nitrification inhibitors, the combination and, finally, cover crops.

4.8 Summary and main findings of the case study

Four GHG mitigation strategies with a focus in corn production, as well as their combination, were assessed for the Typical Farm in Iowa:

(1) The reduction of the nitrogen rate is achieved by splitting the application. The savings in fertilizer are larger than the expenditure of the additional pass. Thus, the strategy generates comparatively low negative costs, except when the nitrogen reduction potential assumed is the lowest (worst case).

(2) Tillage can be reduced to no-till, promoting carbon sequestration and lowering the costs. However, this results in a transitory yield penalty, which triggers ILUC and foregone revenue. No-till amounts to negative costs after the yield penalty is over.

(3) Cover crops promote an increase in the soil organic content and fixate nitrogen, enabling a reduction in the fertilizer rate. Total costs are significantly increased. This strategy allows the highest reduction in GHG emissions of the individual mitigation strategies.

(4) The use of nitrification inhibitors lowers nitrogen losses, which implies that less nitrogen fertilizer is required. The inhibitors are pricier than the savings in fertilizer they generate, increasing costs. The emissions reduction is comparatively low.

(5) All the strategies can be combined without adjustments beyond the changes required in the individual strategies. This implies a significant decrease in emissions but an increase in the costs of both crops, although these are lower than only using cover crops.

Sizable differences in the mitigation potential of these individual strategies were identified. Strategies promoting carbon sequestration offered substantially larger reduction potentials than strategies focused on nitrogen usage, although the effect of carbon sequestration is limited. Depending on the time horizon used, two strategies appear most suitable for the Typical Farm. In the short-term, optimizing the nitrogen rate has the lowest mitigation cost. In the long-run, transitioning to no-till offers the lowest average cost.

5 Case study 2: Brazil – corn in Paraná

Following the analysis from chapter 3.1, Brazil has been selected as one of the case studies due to its relevance in the international agricultural markets as well as the ability to obtain support from local agricultural experts and growers. The support of researchers, especially in the context of the *agri benchmark* network, also was taken into account.

The exchange rate for the national currency, the Brazilian Real, is assumed as 3.44 Reals per USD. This value represents the average annual rate of 2016-2018, published by World Bank (2021). All the calculations are conducted in the local currency and are converted into USD as the last step to avoid the effect of the annual variation in the international exchange rates.

The structure of this chapter and consideration described in this case study follows the same approach as those in chapter 4 for the Typical Farm in Iowa. Therefore, in the cases where repetitions occur or no changes are necessary, the ideas are not explained in detail. Consequently, to fully understand the assessment on this Typical Farm, it is recommended to first read the assessment conducted for the Typical Farm in Iowa.

5.1 Selection of the region

The country is divided into states; thus, most statistical information is provided at least at this level of disaggregation. Consequently, the assessment to determine the location of the Typical Farm of this case study begins at the state level. The same criteria used to choose the country is used to define the region on which the case study focuses.

The climatic conditions in Brazil enable the production of two crops in one growing season. In this system, known as double-cropping, the season can be divided into two. In the case of Brazil, the first one encompasses the period from September until the end of the year; the second one starts in January and finishes in March, although these dates can vary regionally. Compared with the second half, the first half is characterized by better growing conditions, especially higher precipitation, and a lower risk of frost (Braccini et al., 2010).

The most common double-cropping rotation in Brazil is soybean–corn, in which the harvest of soybeans finishes in January and is followed by the seeding of corn directly after (Garcia et al., 2018b). Corn grown in this system is usually referred to as second-season or safrinha. When corn is grown as the main or only crop in the season, it is referred to as summer corn or safra. The yield potential of corn grown as second-season is lower than summer corn due to a shorter growing season. Moreover, rainfall patterns in the second half of the season historically have been less consistent (Spangler et al., 2017), raising the risk associated with growing corn as a second crop and making it more difficult for the grower to estimate the yield. Thus, the grower's inclination to use inputs such as fertilizers is lower as he may perceive a higher risk of not receiving an economic return for his investment. Furthermore, the likeliness of investing in nitrogen fertilization is lower because growers assume that soybeans provide a higher nitrogen credit to corn in double-cropping (Alves et al., 2003).

Additionally, second-season corn's response to nitrogen fertilizer is comparatively small, further promoting the production of the crop without applying nitrogen, a common practice in Brazil (Fuentes et al., 2018). Consequently, it can be inferred that the nitrogen rate is proportionately larger in summer corn than in second-season corn.

The Brazilian National Supply Company (CONAB) is the state company in charge of publishing agricultural statistics. Their database includes the average expenditure to produce selected crops. However, the information is provided aggregated for all fertilizers and only for certain cases - i.e., selected crops in specific regions of the state. Accordingly, not all regions report on summer and second-season corn; some report only one or neither of them. Nonetheless, since this project focuses on areas producing the bulk of the agricultural output, eliminating regions where the crop is not significant is deemed unproblematic.

Still, averaging the available data can be used to indicate an approximation of the expenditure in each production system. Considering the 2016-2018 period for all the regions with information, the average expenditure in fertilizer in second-season corn is 156 USD/ha; in summer corn, 298 USD/ha, or 91% higher (CONAB, 2021b). This difference indicates that the intensity of the production of summer corn is significantly higher, which is in line with what has been previously described. The average yields reported in these regions are 5.5 t/ha and 8 t/ha (or 46% more), correspondingly. Thus, there is an over-proportional expenditure on synthetic fertilizer, revealing that fertilizer use is less efficient in summer corn.

No further considerable differences regarding the production system could be identified in the literature. Therefore, based on the observation that second-season corn receives proportionally lower nitrogen rates and that nitrogen fertilization is a key driver of GHG emissions, it is inferred that the mitigation potential achievable in this system is comparatively low. Thus, the case study is conducted for summer corn, as it is assumed that a larger GHG mitigation potential is achievable in this crop.

For the selection of the region for the case study, the share of the total output of summer corn and yields of various states are used as a first indication. The states with the largest shares of the total output are Rio Grande do Sul (20%), Minas Gerais (19%) and Paraná (13%) (CONAB, 2021a). However, average yields in Paraná (8.6 t/ha) are considerably higher than in Rio Grande do Sul (7.1 t/ha) and Minas Gerais (6.3 t/ha). This suggests that Paraná is the state with the highest intensity of summer corn production. In turn, the intensity of production could be understood as an indicator of the potential to mitigate GHG emissions by adjusting nitrogen fertilization.

The dataset from CONAB (2021b) on regional data is deemed not applicable to make inferences at the state level, as not enough regions are described. The *agri benchmark* Cash Crop network does not have any Typical Farms with summer corn in Brazil either. Thus, it is not feasible to conduct further assessments in this regard. Consequently, based on the data previously presented, the state of Paraná is selected for the case state because it has as a relevant share of the total national production of summer corn and has an intensive production of the crop.

Through members of the network, it was possible to obtain the support of growers and agricultural experts in south-central Paraná. The involvement of these stakeholders is considered essential to derive comprehensive results, so it was decided to conduct the case study there. A new Typical Farm is described following the approach presented in chapter 3.4. For the remainder of this case assessment, the term corn will refer exclusively to summer corn, unless otherwise stated.

5.2 Climate

Table 5.1 presents a summary of temperatures and precipitation. Winter, especially August, is characterized by comparatively less precipitation, although the distribution of the precipitation can be considered homogenous across the entire year. The annual rainfall is 1,866 mm. Beginning in May and until early August, frost is possible, limiting length the production season, although this occurs mostly intermittently (Grodzki et al., 1996).

Table 5.1 Climatic data in Paraná (1976-2005)

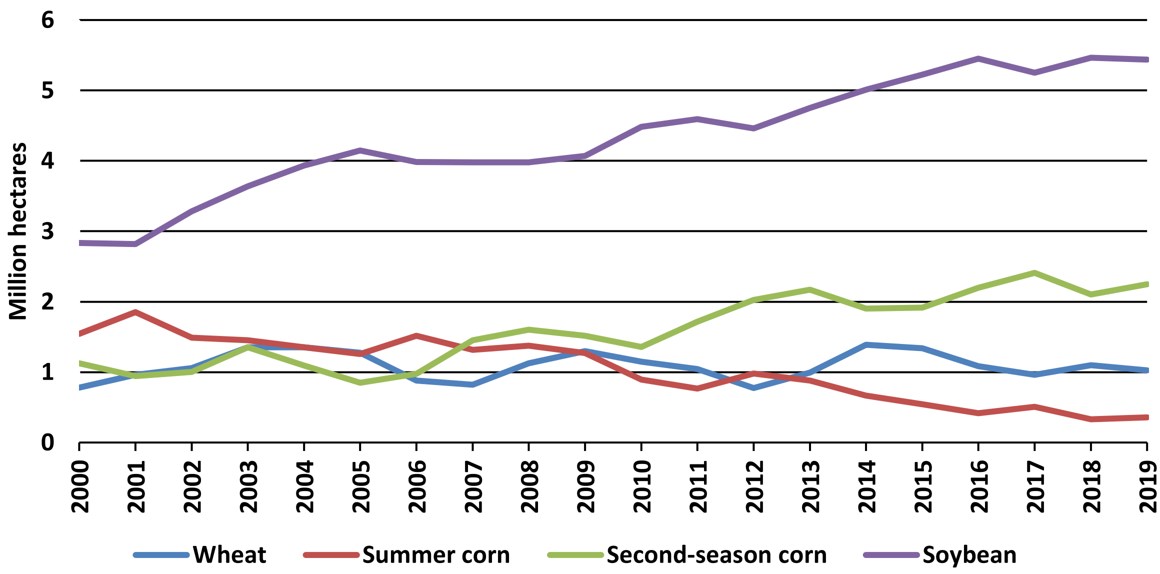
Month	Temperature (°C)	Min. Temp. (°C)	Max. Temp.(°C)	Monthly Rainfall (mm)
Jan	21.7	16.5	26.9	199
Feb	22.6	16.4	26.7	171
Mar	19.8	15.6	26.0	147
Apr	19.7	13.4	23.9	146
May	14.4	10.0	20.7	163
Jun	15.2	8.8	19.6	138
Jul	14.0	8.5	19.5	127
Aug	14.4	9.4	21.4	93
Sep	15.4	10.8	22.0	157
Oct	17.5	13.0	24.0	187
Nov	20.9	14.2	25.5	150
Dec	20.9	15.5	26.3	188

Source:EMBRAPA (2012)

5.3 Background and key features of arable farming in Paraná

According to CONAB (2021a), the total arable land in Paraná (average 2017-2019) is approximately 7.3 million ha, which represents approximately 40% of the state's surface. Soybeans are the most-produced crop, with 74% of the acreage, followed by wheat (14%) and then by summer corn (5%). An additional 2.5 million ha are used for double-cropping, with corn being the most produced second-season crop (2.3 million ha). Thus, the total area harvested per year in the state is 10.3 million ha.

Figure 5.1 Acreage of selected crops in Paraná from 2000 to 2019



Source: own elaboration based on CONAB (2021a)

The evolution in the acreages of these crops is presented in Figure 5.1. Since the year 2000, the acreage of soybeans has increased by 90%. This expansion has occurred mainly through the transformation of pastures into cropping land as well as the clearing of new land. Moreover, a share of the summer corn acreage has switched to soybean-corn double-cropping. The transition has been promoted by considerable improvements in the genetics of second-season corn, which have shortened its growing time, considerably increasing yields (Canalli et al., 2020; Xu et al., 2021). Moreover, soybeans' potential to fixate nitrogen in the soil, thereby reducing the need to use synthetic fertilizers, and its yield potential in colder climates also increased considerably (Alves et al., 2003). Both aspects have contributed toward making double-cropping more profitable than summer corn in various regions of Paraná. However, in other regions of South America with similar agroclimatic conditions, the acreage of summer corn has increased significantly. In northeastern Argentina, which is among the most relevant regions for crop production, the acreage has increased considerably since 2000 (MAGYP, 2021). Thus, while in Paraná the crop may have a downward trend, the crop and mitigation strategies analyzed retain their relevance in the macro-region.

There are predominantly two climatic zones in Paraná, which define the crop rotations. In the subtropical parts of the state (south and southeast), the rotation is based on soybeans, corn, and wheat as annual crops. In the northern, central and western parts of the state, the rotation is based on soybean-corn double-cropping (EMBRAPA, 2011). Additional crops that may be used in the rotation are barley, oats and small legumes.

The most common tillage system in the southern states of Brazil, which include Paraná, is no-till. Ploughing and intensively tilling the soil were recognized as the main promoters of erosion, which led to a significant loss of the topsoil. In the 1970s, growers began adopting no-till as an approach to diminish erosion. No-till is estimated to be used on 92% of the arable acreage in Paraná (EMATER, 2014). Additionally, it is estimated that over 95% of the state's acreage has crops in the winter, with cover crops making up half this acreage (EMBRAPA, 2011).

5.4 Typical Farm and focus group

Since no Typical Farm is available in the region, it was necessary to describe a new one to understand the context in which the mitigation strategies are evaluated. The focus group and the approach used in the discussion are presented in chapter 5.4.2.

5.4.1 Typical Farm and its representativity

Based on the focus group discussion, the Typical Farm is assumed to have 400 ha. Following the latest agricultural census definitions (IBGE, 2018), it belongs in the category of farms with 200 to 500 ha. The farms in this range manage the largest share of land of all the size classes, operating on approximately 20% of Paraná's arable land. Combining the remaining size classes shows that 41% of the state's acreage is managed by smaller farms and 39% by larger size classes. No data is provided in the census regarding yields for the size classes; thus, no inference can be made. Yet, assuming yields are similar between the categories, the Typical Farm could be assumed to belong to the size class producing the majority of agricultural output.

The Typical Farm follows the no-till system, which, as previously presented, is the dominant tillage system in the state. Thus, in this aspect, it is in line with most of the other farms. The three-year crop rotation assumed is: soybean with barley in double-cropping in the first year; soybean in the second year; and lastly, summer corn. Black oats are seeded as a cover crop when double-cropping is not used; that is, after soybean and after corn.

CONAB (2021b) publishes data on the production costs of summer corn in Paraná. As previously mentioned, these are aggregated per input category, such as fertilizer, pesticides, etc. However, these data are based on trials located in a region with a different climatic zone, resulting in noticeable, distinct yields. For the Typical Farm, corn yield is 13 t/ha while in the trials it is only 8.5 t/ha, indicating a difference of more than 50%. Moreover, since climatic conditions partly determine weed and fungi pressure, the application of crop care products may not be directly comparable since the treatments are designed for different contexts. Consequently, the data published is deemed as not comparable to the Typical Farm.

No further information could be identified to evaluate the Typical Farm in the local or regional context.

5.4.2 The Paraná focus group

Scientific literature was used for an initial assessment of the mitigation strategies in the local context preceding the focus group, which took place in April of 2019. Seven growers, an agricultural consultant and the researcher conducting this project participated.

Given that no Typical Farm was available in the region, a draft based on statistical data and selected interviews was created in preparation for the discussion. This preliminary depiction was defined using the period 2016-2018 to describe an average year. After the project and objective had been introduced to the members of the focus group, the draft of the Typical Farm was discussed to realize the necessary adjustments and describe its baseline scenario.

The feasibility of implementing the mitigation strategies and the possible changes in yields, inputs and farm operations were assessed in detail in the focus group. Even though the mitigation strategies focus on corn, the effect on the entire crop rotation was considered in the discussion. Each strategy was discussed until a consensus on the impact was reached. Selected interviews with agricultural experts and visits to local farms were carried out to further evaluate aspects that help understand the context and implications of the strategies. The results derived were compared with scientific literature to validate the data and underlying assumptions.

5.5 Mitigation strategies analyzed

The analysis of the mitigation strategies follows the same approach explained in chapter 4.5 for the Typical Farm in Iowa. Certain aspects assessed of the individual mitigation strategies in the case study of Iowa also are valid for the Paraná Typical Farm. Consequently, in the instances in this case study where the aspects have already been addressed for the Iowa case study, only the key elements pertinent to this farm are examined.

The evaluation of the combination of mitigation strategies for the Paraná Typical Farm is presented in chapter 5.6 and the comparison of the strategies in chapter 5.7. Appendix A contains the tables presenting the assumptions used for the calculation of the mitigation costs in this case study[26].

5.5.1 Status quo – baseline scenario (BR-0)

The baseline scenario of the Typical Farm in Paraná derived from the focus group discussion is presented in this chapter. The mitigation strategies are compared against this baseline scenario to determine the difference in GHG emissions and costs. The costs of corn production for the Typical Farm in Paraná equal 880 USD/ha and are shown in Table 5.4 in chapter 5.5.2. The costs of using black oats as a cover crop are not included but are explained in chapter 5.5.3.

Three tons of dolomitic lime are applied before corn is seeded. However, the effect of this treatment is usually assumed to last for the entire crop rotation on the Typical Farm, as mentioned by the focus group. Thus, although the application occurs in this crop, all the other crops benefit as well. Therefore, for purposes

[26] Input costs are presented in Table A.1, coefficients for the GHG calculation in Table A.2, inputs used in corn in Table A.9, and farm operations in Table A.13.

of the costs and GHG estimations, it is assumed corn received only 1 ton of lime, which is already included in the cost previously indicated.

The *agri benchmark* network does not have a Typical Farm with corn in the region, thus the costs of machinery are depicted as zero, although logically, this is not the real case. This does not affect the estimation of the mitigation costs, as only the difference between the strategies and the baseline scenario are required to determine the costs of the mitigation strategy. No contractor services are used in the Typical Farm; thus, the value is zero.

Emissions per hectare in the baseline scenario

The GHG emissions from the baseline scenario consider only the production of corn and do not include the emissions from black oats as cover crops. For the estimation of the indirect losses, the adjustment factor must be calculated. This calculation considers all the nitrogen applications in corn. The previous crop is soybeans. Alves et al. (2006) mention that in the region of the Typical Farm, soybeans' fixation generates a nitrogen surplus that can be taken by the following crop. Still, the research by Siqueira Neto et al. (2010) indicates that a significant share of the nitrogen fixated can be lost between the harvest of soybeans and the seeding of a summer crop. Thus, while second-season corn can benefit from this nitrogen credit as it is grown directly after soybeans, summer corn may not benefit due to the losses in the time between harvest and seeding. In this context, no scientific data on nitrogen availability could be identified for the region. Nonetheless, members of the focus group mentioned that previous soil samples taken before summer corn is seeded have indicated low or negligible amounts of nitrogen from the soybeans' fixation. Consequently, they do not consider any nitrogen credit toward summer corn. The same assumption is used in this analysis. As a result, the input of nitrogen in corn is assumed to be provided only by synthetic nitrogen fertilizers, which are urea (189 kg N/ha) and monoammonium phosphate (MAP) (32 kg N/ha).

To calculate the nitrogen removal via harvest of corn, the moisture content of the grain is assumed to be 14% based on the focus group and the nitrogen content of the grain as 1.67% of the dry matter[27] (Garcia et al., 2018a). Therefore, 176 kg N/ha are removed via harvest. The same moisture and nitrogen content of the grain are assumed for all the mitigation strategies in this Typical Farm.

Based on the maps provided by Sayre et al. (2020), it is assumed that the region has a Warm Temperate Moist Climate, which is used for the estimation of emissions based on IPCC (2019a). Table 5.2 presents the emissions of the Typical Farm in the baseline scenario. A summary of the emissions entailed in each category is provided in chapter 3.5.7.

[27] Garcia et al. (2018a) provide the nitrogen content of the corn grain based on various fertilizer treatments based on urea. The value used is the average of all the treatments presented on the first site, which is deemed a better representation of the soil present on the Typical Farm in Paraná.

Table 5.2 GHG emissions from corn for Typical Farm in Paraná - BR – 0

	Hectare ($kg\ CO_{2eq}/ha$)	Product ($kg\ CO_{2eq}/t$ crop)
Direct	1,957	151
Indirect	226	17
Liming	551	42
Urea	301	23
Manufacture	602	46
Diesel	63	5
Land use	0	0
Total	**3,700**	**285**

Source: own estimation.

The total GHG emissions of corn in the baseline scenario are 3,700 kg CO_{2eq}/ha. The Typical Farm has been operated with the same tillage regime and input of biomass for longer than 20 years. Consequently, no net change in the carbon content of the soil is assumed (see chapter 3.5.6).

5.5.2 Optimization of nitrogen rate (BR-1)

The GHG emissions from corn production could be reduced by optimizing nitrogen fertilization by increasing the number of passes to spread fertilizer and utilizing a lower total rate, although the feasibility must be assessed. The potential to lower the nitrogen rate can be obtained by comparing the current NUE with optimal values from the literature. The difference between them reveals the extent to which the rate can be reduced without theoretically affecting yields. In the baseline scenario, the removal is estimated to be 176 kg N/ha and the nitrogen input as 221 kg N/ha. This results in a NUE of 79%. The optimal values for NUE from Brentrup and Palliere (2010) are used in this Typical Farm to calculate the reduction potential as no region-specific coefficients could be identified (see chapter 3.6.2.1). The resulting nitrogen fertilization rate in the standard, best and worst cases is presented in Table 5.3.

Table 5.3 Reduction of nitrogen rates based on optimal NUE for Typical Farm in Paraná

Case	N removed via harvest	Fertilization rate	NUE	N reduction potential
	(kg N/ha)	(kg N/ha)	(%)	(kg N/ha)
Baseline	176	221	79	-
Standard	176	207	85	16
Best	176	195	90	27
Worst	176	219	80	3

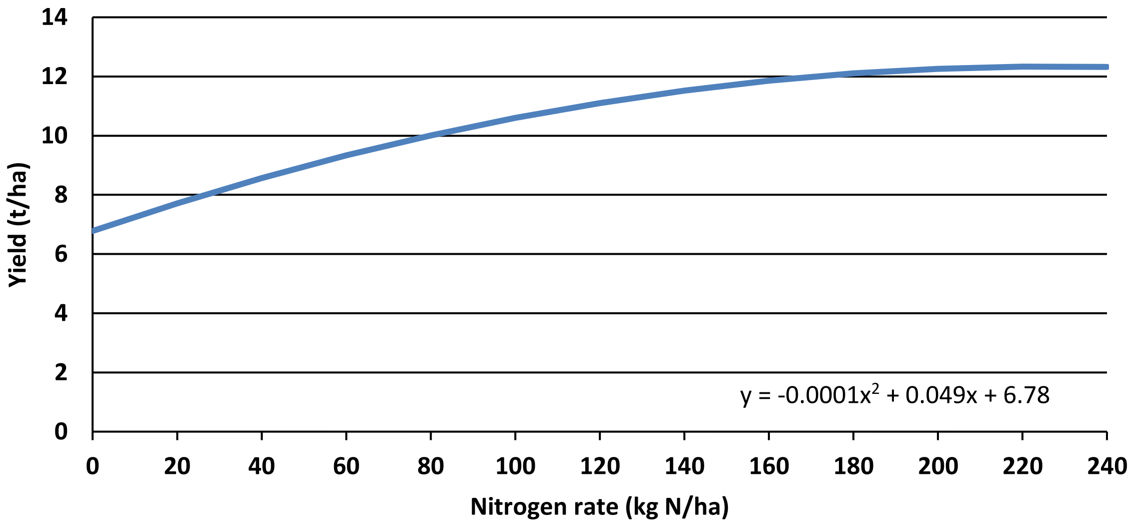
Source: own estimation based on Brentrup and Palliere (2010).

The NUE approach used in calculating the potential to reduce the nitrogen rate is not specific to the region of this case study. Thus, it is necessary to evaluate the possible effect nitrogen reduction may have on yields utilizing data from the region selected or one that presents comparable conditions. Thus, to verify the potential effect of the reductions on yields, the nitrogen response curve from Fontoura and Bayer (2009) is used (see Figure 5.2). The curve is based on data from several average-yielding fields with climatic conditions similar to those of the Typical Farm. The regression indicates a maximum yield of 12.8 t/ha, which is slightly lower than the Typical Farm's yield. However, this difference can be attributed to yield improvements from plant breeding, as the author's estimations are based on results from the period 2005-

2007. Consequently, while the curve may not exactly represent the current situation, it is deemed a valid approximation.

To calculate the change in the yield derived from nitrogen reduction, the nitrogen rate in the baseline scenario and the reduced rates are applied in the formula to calculate the possible change in the yields. Thus, according to Fontoura and Bayer's (2009) regression, the yield reductions from the mitigation strategy compared with the baseline scenario are: 0.03% in the worst case; 0.3% in the standard case, and 0.9% in the best case. These reductions are deemed insignificant; thus, lowering the nitrogen rate is assumed to not affect the yields in the Typical Farm.

Figure 5.2 Estimated corn yield under different nitrogen fertilization rates in Paraná



Source: own estimation based on Fontoura and Bayer (2009).

In the baseline scenario, the application of nitrogen occurs at seeding in mid-September using MAP (32 kg N/ha) and three weeks later as urea (189 kg N/ha). The reduction calculated must be discounted to urea. Lowering the MAP rate implies reducing the input of phosphate, which is also contained in the fertilizer. The lacking amount of phosphate would need to be supplied as a different fertilizer, which would need to be manufactured, releasing emissions, or could pose further agronomic challenges. Moreover, the emissions entailed in the manufacture of one kg of nitrogen as urea are higher than as MAP (Brentrup et al., 2018). Therefore, lowering the urea rate would achieve greater mitigation. Thus, the rate of urea is reduced as presented in Table 5.3. To apply this reduced urea rate, an additional pass is assumed, splitting the original urea application into two. Each pass contains the same amount of urea. The additional pass was deemed necessary by the participants of the focus group as a strategy to reduce losses (see chapter 3.6.2.1).

The focus group stated that soybean seeding begins directly after the application of urea is complete (mid-October) and lasts until the end of November. Hence, a theoretical additional application of urea in this period would delay seeding. Applying after the seeding is done was deemed problematic by the participants of the focus group as December marks the beginning of the rainy season, limiting the possibility to operate machinery without risk of compacting the soil. Moreover, the crop's size at that stage also may pose a problem, as not all tractors and equipment are high enough to operate without damaging the plant. Thus, based on the focus group consensus, the use of agricultural contractors was deemed as the preferred option for the additional pass. The cost of the application by the contractor is assumed to be 10 USD/ha, and requires 1.5 l/ha of diesel, which is accounted for in the estimation of emissions.

The additional operation to spread urea is assumed to be conducted in mid-November, approximately two months after corn has been seeded. Panison et al. (2019) demonstrate that splitting nitrogen applications in corn, applying until approximately 75 days after seeding, does not influence yields compared with single applications of the fertilizer. Hence, no change in yields is assumed from splitting the fertilizer rate.

Total costs from implementing the mitigation strategy

The changes in costs from optimizing nitrogen fertilization are presented in Table 5.4. Lowering the rate reduces the expenditure on fertilizer depending on the reduction achievable. The reduction in the standard case is -13 USD/ha. In the best case, the change amounts to -22 USD/ha and in the worst case, only -2 USD/ha. The costs of a contractor are increased by 10 USD/ha in all cases, as this fee is not altered by the changes in the fertilizer rate. None of the other cost categories change compared with the baseline scenario.

In the standard and best cases, the mitigation strategies generate a win-win scenario. The total changes result in a reduction of 3 and 12 USD/ha, respectively. The strategy entails an increase of 8 USD/ha when the nitrogen reduction potential is assumed to be the lowest - that is, the worst case.

Table 5.4 Corn costs with optimized nitrogen fertilization rate for Typical Farm in Paraná (USD/ha)

	Baseline scenario	Standard case		Best case		Worst case	
		Strategy	Change	Strategy	Change	Strategy	Change
Seed	261	261	0	261	0	261	0
Fertilizer	393	380	-13	371	-22	391	-2
Pesticides	163	163	0	163	0	163	0
Liming	34	34	0	34	0	34	0
Diesel	19	19	0	19	0	19	0
Labor	10	10	0	10	0	10	0
Machinery	0	0	0	0	0	0	0
Contractor	0	10	10	10	10	10	10
Total	**880**	**877**	**-3**	**868**	**-12**	**888**	**8**

Source: own estimation.

Emissions per hectare from implementing the mitigation strategy

Table 5.5 shows the change in GHG emissions from optimizing the nitrogen rate. The N_2O emissions from direct and indirect are lowered depending on the nitrogen reduction. In the standard case the reduction is estimated to be 201 kg CO_{2eq}/ha. In the best case it is calculated as 350 kg CO_{2eq}/ha and in the worst case only 34 kg CO_{2eq}/ha. Comparably, the emission from urea and manufacture of fertilizer are lowered by varying magnitudes, as these also depend on the input of fertilizers. The emissions from the sum of these two categories range from -101 to 10 kg CO_{2eq}/ha, in the best and worst cases, with a value of 59 kg CO_{2eq}/ha in the standard case.

The diesel emissions resulting from the additional pass are 5 kg CO_{2eq}/ha in the three cases. Yet, compared with the reductions achieved from the lowered fertilizer rate, this increase is comparatively low. Liming and land use are grouped in the category “remaining,” as they are not affected by the implementation of the GHG mitigation strategy.

The total change in the standard case is a reduction of 255 kg CO_{2eq}/ha; 446 kg CO_{2eq}/ha in the best case; and 39 kg CO_{2eq}/ha in the worst case.

Table 5.5 GHG emissions from corn with optimized nitrogen fertilization rate for Typical Farm in Paraná (kg CO_{2eq}/ha)

	Baseline scenario	Standard case		Best case		Worst case	
		Strategy	Change	Strategy	Change	Strategy	Change
Direct	1,957	1,841	-116	1,755	-202	1,938	-19
Indirect	226	141	-85	79	-148	212	-14
Urea	301	276	-25	258	-43	297	-4
Manufacture	602	568	-34	544	-59	597	-6
Diesel	63	67	5	67	5	67	5
Remaining	551	551	0	551	0	551	0
Total	**3,700**	**3,445**	**-255**	**3,254**	**-446**	**3,661**	**-39**

Source: own estimation.

Mitigation costs and considerations on the adoption, monitoring and enforcement of the strategy

The resulting mitigation costs are: -10 USD/t CO_{2eq} in the standard case; -26 USD/t CO_{2eq} in the best case; and 206 USD/t CO_{2eq} in the worst case. Thus, the mitigation strategy represents a win-win in two of the three scenarios, as the grower would decrease its costs as well as the emission (see chapter 8).

The considerations on the adoption, monitoring and enforcement of this mitigation strategy already have been presented for the case study of Iowa in chapter 4.5.2 and are comparable in this case study.

5.5.3 Legumes as cover crops (BR-2)

Theoretically, using cover crops in the rotation enables the sequestration of carbon because they increase the crop residues that can be incorporated into the soil (see chapter 4.5.4). However, the Typical Farm in Paraná already uses black oats as a cover crop in the baseline scenario. Thus, to bind additional carbon in the soil, the amount of biomass generated that is incorporated must be increased. Tiecher et al. (2020) tested seven common cover crops in Paraná over 26 years in a corn and soybean rotation with management and climatic conditions comparable to the Typical Farm. They concluded that black oats yield the most biomass given the local context, producing approximately 4.9 t/ha of above-ground dry mass. Hence, using a different cover crop likely would reduce the input of residues because of the reduced biomass generated, releasing the carbon previously sequestered. Therefore, replacing black oats is deemed not viable to increase soil organic carbon.

Hypothetically, fertilizing the cover crops would increase the biomass they produce, promoting carbon sequestration, but the manufacture and usage of fertilizer imply additional emissions. Amado et al. (2003) indicate black oats as cover crops produce approximately 1.1 t/ha more dry biomass when fertilized with 40 kg N/ha. The biomass can be assumed to be 42% carbon (Mani et al., 2011) and approximately 10% of this carbon remains sequestered in the soil after 20 years (Sørensen, 1987). Assuming a conversion factor from carbon to CO_2 of 3.7 (IPCC, 2019a), it can be calculated that each kg of nitrogen used binds 4 kg CO_{2eq}/ha in the soil. However, the GHG emissions triggered by the manufacture and application of nitrogen are approximately 10 kg CO_{2eq}/ha. Thus, it can be argued that the short-term balance of fertilizing black oats is a net increase in emissions. Therefore, it is not considered a viable approach to sequester carbon.

The focus group mentioned the possibility of using a mixture of seeds to include a legume to fixate nitrogen and lower the rate of synthetic nitrogen for the following crop. They added that the most common legume used as cover crop in Paraná is common vetch. Tiecher et al. (2020), whose work already was discussed, show that vetch has the highest biomass yield from the legumes tested. Participants of the focus group stated that they have obtained similar amounts of biomass using a two-to-one vetch-oat mixture compared with oats alone, both with the same seeding rate of 80 kg/ha. Giacomini et al. (2003) tested various ratios of the vetch-oat mixture utilizing the same seed rate. The authors concluded that using a mix of 67% vetch and 33% oats yields the same biomass as using only oats, confirming the statement from the participants. Thus, a mixture of 67% vetch and 33% oats is assumed for the Typical Farm, maintaining the same total seeding rate as in the baseline scenario (80 kg/ha). The cover crop seeds are available in the market as these crops are commonly used. No applicable data on the input and management of the production of the seeds for the cover crops could be identified. However, it could be expected that the production of vetch seeds releases fewer emissions than oats, as legumes usually are not fertilized with nitrogen. Thus, by including vetch in the mixture, the overall emissions from the seeds for cover crops are arguably lower. Nonetheless, this benefit cannot be corroborated.

As in the baseline scenario, the oat seeds are assumed to be farm-saved and to cost 0.2 USD/kg, as indicated by the focus group participants. In the case of common vetch, they mentioned having problems with the germination of farm-saved seeds. Thus, they indicated they utilize a mixture of 67% certified (0.9 USD/kg) and 33% farm-saved (0.3 USD/kg), which indicates a cost of 0.7 USD/kg. Assuming seeding rates of 27 kg/ha for oats and 53 kg/ha for vetch, the total cost of seeds for the mixture is 42 USD/ha.

Moreover, the focus group commented on the need to inoculate the vetch seeds to promote biological nitrogen fixation. This treatment costs 7 USD/ha, hence the seed-related costs in this mitigation strategy total 49 USD/ha. The 80 kg/ha of oat seeds applied in the baseline scenario cost 17 USD/ha. The net difference in costs between the mitigation strategy and the baseline scenario (32 USD/ha) is added to the seed cost of corn.

The mixture of seeds can be spread with the broadcaster as in the baseline scenario. The focus group mentioned that the machinery could be used without significant adjustments. Similarly, the termination of the cover crop with glyphosate in the baseline scenario is assumed for the mixture. Thus, no additional costs are generated from seeding or terminating the mixture. However, they mentioned that the addition of vetch requires spraying insecticide on the cover crop. This additional application was considered necessary because, otherwise, pests could severely damage vetch in the early stages, lowering the biomass production and fixation potential significantly. No competition for machinery occurs, as no other operations take place at the same time. The additional pass, assumed to be conducted with the grower's own equipment, increases pesticide costs by 12 USD/ha, as well as labor, diesel and maintenance and repairs[28]. No additional depreciation occurs, as the machinery is assumed to still operate within the expected annual devaluation.

Black oats also are used as a cover crop after corn that is followed by soybeans in double-cropping. Since soybeans are legumes, no synthetic nitrogen fertilizer is applied. Hence, using the mixture as opposed to only black oats would increase the expenditure but would not generate a reduction in emissions, as no nitrogen is used.

The nitrogen credit from the mixture toward corn was estimated to be between 30 and 40 kg N/ha by the growers of the focus group. Amado et al. (1998) calculated the nitrogen contribution of an oat–vetch mixture toward corn as 38 kg N/ha. Nevertheless, the exact ratio of the seeds, nor the rate, are reported. Giacomini et al. (2004) seeded 80 kg/ha of a mixture composed of vetch–oat in a ratio of 70-30 and measured a nitrogen credit between 36 and 42 kg N/ha. They add that the residues are completely disintegrated at the end of the season. Both these estimations were obtained in southern Brazil in

[28] The costs of repairs were stated by the participants of the focus group. Thus, it is not derived from the approach explained in chapter 3.4.3.

conditions similar to the Typical Farm and are comparable with the approximation provided by the focus group. Thus, the range indicated by the focus group is assumed as the nitrogen provided by the mixture toward corn. The values used for the standard, best and worst cases are shown in Table 5.6.

Table 5.6 Nitrogen rate and supply from fixation by cover crop for Typical Farm in Paraná

Case	Nitrogen from fixation (kg N/ha)	Synthetic nitrogen (kg N/ha)	Total nitrogen supply (kg N/ha)
Standard	35	186	221
Best	40	181	221
Worst	30	191	221

Source: own estimation.

The total nitrogen rate for corn remains unchanged and the supply from the cover crops is assumed to reduce the rate of urea. The above-ground biomass of the mixture is assumed to be 4.9 t/ha (Tiecher et al., 2020) and is completely decomposed by the end of the season (Bortolini et al., 2000). To determine the nitrogen content in the crop residues, the biomass is assumed as 60% oats and 40% vetch (Giacomini et al., 2003). For the estimation of emissions from residues, the difference between the nitrogen content in the cover crops in the baseline scenario and in the mixture is added to the corn residues[29].

Total costs from implementing the mitigation strategy

Table 5.7 presents the costs of using the mixture vetch–oat as cover crop, replacing the original of only oats. The increase in seed costs is the result of changing the composition of the seed mixture and using the inoculant, which is included in this category. The increase totals 32 USD/ha and is the same in the three cases. The expenditure in fertilizer is lowered by 29 USD/ha in the standard case, by 33 USD/ha in the best case and by 25 USD/ha in the worst case. The difference is explained by the varying amounts of nitrogen that vetch is assumed to provide toward corn.

The expenditures in pesticides, diesel, labor and machinery increase as a result of the additional application of insecticides. The category "machinery" has an increase of 0.3 USD/ha from additional repairs and maintenance yet is not displayed in the table because it is rounded down to zero. The increase in these categories is same the same in the three cases, resulting in an increase of 13 USD/ha.

Liming and contractor are contained in the category "remaining," as these are not affected by the mitigation strategy. The total increase in corn is a net increase of 16 USD/ha in the standard case, 12 USD/ha in the best case and 21 USD/ha in the worst case.

[29] The standard values from IPCC (2019a) are used to determine the nitrogen addition from crop residues and are presented in Table A.5. The values from Tiecher et al. (2020) are assumed to depict the above-ground dry mass (see chapter 3.5.1.1). Thus, the residues of oats are calculated to contain a total of 39 kg N/ha and the mixture, 96 kg N/ha. The difference of 57 kg N/ha is added to corn crop residues.

Table 5.7 Costs in corn with legumes in cover crop mixture for Typical Farm in Paraná (USD/ha)

	Baseline scenario	Standard case		Best case		Worst case	
		Strategy	Change	Strategy	Change	Strategy	Change
Seed	261	293	32	293	32	293	32
Fertilizer	393	364	-29	360	-33	368	-25
Pesticides	163	174	12	174	12	174	12
Diesel	19	20	1	20	1	20	1
Labor	10	11	1	11	1	11	1
Remaining	34	34	0	34	0	34	0
Total	**880**	**896**	**16**	**892**	**12**	**900**	**21**

Source: own estimation.

Emissions per hectare from implementing the mitigation strategy

The GHG mitigation achieved with the mixture of seeds as cover crops is presented in Table 5.8. The emissions from direct, which are driven by the nitrogen applied and contained in the crop residues, are decreased by 102 kg CO_{2eq}/ha in the standard case. In the best case, the decrease is 140 kg CO_{2eq}/ha and in the worst case, 65 kg CO_{2eq}/ha. The difference is explained by the fixation potential attainable, which decreases the rate of synthetic nitrogen. The indirect emissions are similar in magnitude in the three cases, ranging from -1 to -2 kg CO_{2eq}/ha.

The different nitrogen fixation rates from vetch also explain the variation in the reduction in emissions from urea and manufacture, as they are directly linked to the use of the fertilizer. The sum of both categories results in a reduction of 133 kg CO_{2eq}/ha in the standard case, 152 kg CO_{2eq}/ha in the best case and 114 kg CO_{2eq}/ha in the worst case. Diesel emissions increase by 3 kg CO_{2eq}/ha in all cases, as the same pesticide application is assumed. Compared with the reductions achieved in the other categories, this increase can be considered insignificant. The emissions from lime application are not affected and are included in the category “remaining.”

The total emissions with the mitigation strategy are a reduction of 234 kg CO_{2eq}/ha in the standard case, 290 kg CO_{2eq}/ha in the best case and 178 kg CO_{2eq}/ha in the worst case.

Table 5.8 GHG emissions, corn with legumes in cover crop mixture for Typical Farm in Paraná (kg CO_{2eq}/ha)

	Baseline scenario	Standard case		Best case		Worst case	
		Strategy	Change	Strategy	Change	Strategy	Change
Direct	1,957	1,855	-102	1,818	-140	1,893	-65
Indirect	226	225	-2	225	-1	224	-2
Urea	301	245	-56	237	-64	253	-48
Manufacture	602	526	-77	515	-88	537	-66
Diesel	63	65	3	65	3	65	3
Remaining	551	551	0	551	0	551	0
Total	**3,700**	**3,466**	**-234**	**3,410**	**-290**	**3,522**	**-178**

Source: own estimation.

Mitigation costs and considerations on the adoption, monitoring and enforcement of the strategy

The cost to mitigate emissions with legumes in the mixture of cover crops is 70 USD/t CO_{2eq} in the standard case, 42 USD/t CO_{2eq} in the best case and 115 USD/t CO_{2eq} in the worst case.

Monitoring and enforcing the use of legumes as cover crops could be achieved via remote sensing. However, given that cover crops already are used in the baseline scenario, this approach would imply additional challenges compared with a case in which cover crops are not used in the baseline scenario. Satellite data would need to be analyzed to identify the individual plants growing at the same time. In many cases, computational models are not yet precise enough to fulfill this task, but these are actively being developed. Hence, they could potentially distinguish a variety of different plant species within the same plot (Wang and Gamon, 2019). Thus, assuming the identification of the plants in the mixture becomes feasible, monitoring the adoption of the mixture could be realized on a wide scale in a cost-effective method. However, until this approach is feasible, monitoring would be comparatively more expensive, as on-farm visits would likely be required to determine adoption of the mixture.

Alternatively, establishing a tax on nitrogen would make the cost of this strategy comparatively cheaper, as the reduction of urea due to the nitrogen fixation would result in higher economic savings. This possibility was discussed in chapter 4.5.4.

5.5.4 Enhanced-efficiency fertilizer - urease inhibitors (BR-3)

The availability of nitrogen for the crops depends on the fertilizer type, soil and weather, among other factors. In the Typical Farm in Paraná, a significant share of the nitrogen supplied to corn is as urea. Compared with other forms of nitrogen fertilizer, urea undergoes several chemical changes in the soil before the nutrient becomes available to the plant. Throughout these transformations, nitrogen leaches and/or is lost to the environment as gas (see chapter 3.6.2.4). Chemical additives can be used to reduce the transformation rates and promote a reduction of losses. As a result, the urea rate theoretically could be reduced without affecting the availability of the crop because the losses to the environment would decrease.

Various additives were presented and discussed in the focus group. The urease inhibitor NBPT was regarded as the most feasible alternative for the Typical Farm. This chemical compound reduces the activity of the enzymes responsible for the initial transformation of urea in the soil, lowering the losses as NH_3 (Cantarella et al., 2018). Reducing these losses also decreases N_2O emissions from deposition, entailed in the emissions category “indirect.” The participants of the focus group added that NBPT is the only alternative commercially available that they are aware of in the region and have tested in the past. This urease inhibitor is the most commonly used and studied in Brazil (Modolo et al., 2018; Viero et al., 2017; Cantarella et al., 2008).

Fontoura and Bayer (2010) measured the volatilization losses from 150 kg N/ha of urea treated with NBPT applied to corn in the region of the Typical Farm. The average of the four years reported by the authors reveals that the urease inhibitor decreased NH_3 losses by 47% compared with untreated urea, with a range between 42% and 58%. Viero et al. (2014) report a reduction of 53% to 71% for a similar region in analogous cropping conditions, also applying 150 kg N/ha. The authors of both studies indicate that corn yields were not affected by the use of the NBPT. In a meta-analysis covering multiple regions and crops, Silva et al. (2017) found that NBPT lowered NH_3 losses by an average of 57% across all studies, with a range of 53% to 61%. Arguably, the values provided by the authors of these three studies are similar in magnitude. Taking into account that the trials from Fontoura and Bayer (2010) are from the same region and crop as the Typical Farm, their results are used. Thus, it is assumed that the NH_3 losses from urea are lowered by 47% in the standard case, 58% in the best case and 42% in the worst case.

Theoretically, inhibitors can affect the nitrogen losses from other pathways as well. In this regard, NBPT has been shown to not cause a statistically significant change in the release of N_2O (Akiyama et al., 2010). Similarly, NO_3 leaching can be assumed to not be affected by the use of this urease inhibitor (Rech et al., 2017; Sanz-Cobena et al., 2017; Dougherty et al., 2016; Gioacchini et al., 2002). These studies assessing leaching and N_2O losses depict a variety of crops, nitrogen fertilizers and agroclimatic conditions, as regional data corresponding to the Typical Farm could not be identified. Nevertheless, these are deemed a valid indication of the likely effect of NBPT on these losses in Paraná. Consequently, no other changes in the nitrogen losses are assumed.

To calculate the amount of urea that can be reduced, the volatilization losses in the baseline scenario (BR-0) are taken as a reference value. These assume a volatilization rate of 15%, as indicated by IPCC (2019a). Nonetheless, this coefficient depicts the sum of the losses as NH_3 and NO_x (see chapter 3.5.1.3). According to IPCC (2019a), 93% of the volatilization losses occur as NH_3. Consequently, the losses of the baseline scenario are reduced by 7%. Lastly, this value is modified using the adjustment factor for indirect losses of 0.4 (see chapter 3.5.1.2) to calculate the urea losses as NH_3. This coefficient is multiplied by the share reduced based on Fontoura and Bayer (2010), as previously explained. The result is assumed to indicate the reduction in the urea rate achievable without affecting yields (see Table 5.9).

Table 5.9 Effect of urease inhibitor on volatilization and fertilizer reduction for the Typical Farm in Paraná

Scenario	Urea rate (kg N/ha)	Losses as ammonia (kg N/ha)	Share reduced (%)	Urea reduction (kg N/ha)
Standard	189	10	47	5
Best case	189	10	58	6
Worst case	189	10	42	4

Source: own estimation.

Urea can be purchased already treated with NBPT, as indicated by the focus group, which costs on average 10% more than conventional urea. Based on previous experiences from the growers, the fertilizer broadcaster does not require additional adjustments compared with common urea, nor are changes to the application procedure necessary. Consequently, it can be quickly adopted for all the urea applications in corn without altering the expenditure on diesel, machinery or labor.

Total costs from implementing the mitigation strategy

Utilizing urea treated with NBPT affects only the expenditure on fertilizer. All other costs are as in the baseline scenario (BR-0), which are presented in Table 5.4. The urease inhibitor reduces the rate needed in corn; yet it raises the cost per unit of fertilizer. The total increase in costs in fertilizer in the standard and best cases is approximately 11 USD/ha and in the worst case it is 12 USD/ha. The minimal cost difference is a consequence of the small variation in the reduction of the urea rate. Since no other cost differs, this increase in the fertilizer costs represents the total cost of the mitigation strategy per hectare.

Emissions per hectare from implementing the mitigation strategy

Table 5.10 presents the change in GHG emissions from using urease inhibitor to reduce the urea rate. The reduced fertilizer input lowers the emissions from direct by 39 kg CO_{2eq}/ha in the standard case, 46 kg CO_{2eq}/ha in the best case and 35 kg CO_{2eq}/ha in the worst case. The reduced rate also lowers indirect emissions by 28 kg CO_{2eq}/ha, 34 kg CO_{2eq}/ha and 26 kg CO_{2eq}/ha in the standard, best and worst cases, respectively.

The emissions from the category “urea,” which depict the CO_2 released from the chemical structure of the fertilizer, are lowered as well. The effect varies between a reduction of 7 and 9 kg CO_{2eq}/ha depending on the case. Similarly, the lowered urea rate affects the carbon footprint entailed in manufacture. The reduction, in this case, is 10 kg CO_{2eq}/ha in the standard case, 13 kg CO_{2eq}/ha in the best case and 9 kg CO_{2eq}/ha in the worst case.

The category “remaining” contains the categories liming, diesel and land use, which are not affected by the mitigation strategy.

The net change from using the urease inhibitor in the Typical Farm in Paraná is a reduction of 85 kg CO_{2eq}/ha in the standard case, 102 kg CO_{2eq}/ha in the best case and 77 kg CO_{2eq}/ha in the worst case. These are the values used to calculate the mitigation cost per ton of CO_2.

Table 5.10 GHG emissions in corn using urease inhibitor for Typical Farm in Paraná (kg CO_{2eq}/ha)

	Baseline scenario	Standard case		Best case		Worst case	
		Strategy	**Change**	**Strategy**	**Change**	**Strategy**	**Change**
Direct	1,957	1,918	-39	1,911	-46	1,922	-35
Indirect	226	198	-28	192	-34	200	-26
Urea	301	293	-7	292	-9	294	-7
Manufacture	602	592	-10	590	-13	593	-9
Remaining	613	613	0	613	0	613	0
Total	**3,700**	**3,615**	**-85**	**3,598**	**-102**	**3,623**	**-77**

Source: own estimation.

Mitigation costs and considerations on the adoption, monitoring and enforcement of the strategy

The mitigation cost achieved using urease inhibitors is 134 USD/t CO_{2eq} in the standard case. In the best case, the cost is 103 USD/t CO_{2eq} and in the worst case, 155 USD/t CO_{2eq}.

The adoption, monitoring and enforcement of this type of mitigation strategy has been assessed previously in chapter 4.5.5.

5.6 Combination of strategies (BR-C)

The explanation and approach utilized for the combination of strategies in the Typical Farm in Paraná follow the same principles previously explained for the case study in Iowa in chapter 4.6. The evaluation of the possibility to combine the strategies in the Typical Farm in Paraná has the baseline scenario (BR-0) as its reference.

None of the mitigation strategies assessed in this case study require interrupting or replacing any operation. The optimization of nitrogen fertilization (BR-1) and the use of legumes in the cover crop mixture (BR-2) each entail an additional operation. The optimization implies an additional pass to spread urea during the growing season and is carried out by contractors. The mixture of cover crops requires an additional application of pesticides. This operation is conducted after the growing season and assumes the grower’s machinery is used, yet it does not compete with any other operation. Therefore, in terms of labor and machinery, no competition results from the joint implementation of these two strategies. The use of urea treated with urease inhibitors (BR-3) requires no changes in scheduling or mechanization by the producer. Hence, it does not impose limitations compared with the baseline either.

None of the single strategies imply changes in soil or crop management; i.e., the residues from corn and the cover crops are the same as in the baseline scenario. Therefore, it is presumed that no limitations result from these elements when combining the three strategies. No additional limiting factors affecting yields or the production system are identified. Consequently, it is feasible to implement the three strategies in combination for corn production.

The assessment of the resulting combination is presented in the following chapter. As with the previous combination assessed, only the elements from the individual strategies that may cause interactions are evaluated; assumptions covered in the individual strategies that remain unchanged in the combination are not discussed again.

The baseline scenario is used as the reference to compare the changes in costs and GHG emissions. The standard, best and worst cases of each strategy are matched with the corresponding case of the other strategies.

Costs and mitigation potential of optimization of nitrogen rate, use of legumes as cover crops and urease inhibitors combined

Each of the individual mitigation strategies alters the nitrogen fertilizer rate. Thus, possible interactions and their effects must be evaluated to calculate the total change in fertilizer rate and farm operations.

Following the NUE approach, the optimization of the nitrogen rate (BR-1) diminishes the nitrogen rate to the amount required to obtain the corn yield. Thus, nitrogen losses from the soil are minimized. The urease inhibitor NBPT (BR-3) reduces the rate at which the nitrogen is lost from the system, increasing the share of the nitrogen applied that is available for the crop, thereby enabling the use of a nitrogen rate lower than the indicated by the optimal NUE. Furthermore, the nitrogen fixation from common vetch, included in the cover crop mixture (BR-2), provides another source of nitrogen, replacing a share of the rate applied as synthetic fertilizer. The combination of these three strategies was already described in detail in chapter 4.6. The same methodology to calculate the resulting fertilizer rate is applied in this case study.

The supply of nitrogen from the cover crop mixture can be assumed to not be affected by the use of NBPT (Banerjee et al., 1999). Thus, to determine the rate, first, nitrogen fixated by the legumes (BR-2) is subtracted from the optimized nitrogen rate (BR-1) to depict the rate of synthetic fertilizer. The emissions released by this calculated rate are utilized as the reference to determine the reduction in the losses from the urease inhibitor (BR-3). In turn, the losses reduced by the inhibitor are subtracted from the fertilizer rate. The values are presented in Table B.5 in the Appendix. The total combined reduction in Table 5.11 decreases the nitrogen applied as urea; MAP application is assumed unchanged.

Table 5.11 Nitrogen rate in combination of strategies for Typical Farm in Paraná

Case	Rate in Baseline (BR-0) (kg N/ha)	Reduction of nitrogen rate: Optimization of nitrogen (BR-1) (kg N/ha)	Reduction of nitrogen rate: Cover crop (BR-2) (kg N/ha)	Reduction of nitrogen rate: Urease inhibitors (BR-3) (kg N/ha)	Rate in Combination (BR-C) (kg N/ha)
Standard	221	16	35	2	168
Best case	221	27	40	1	152
Worst case	221	3	30	3	184

Source: own estimation.

The limitations on assessing the effect of the combined strategies and the nitrogen rate have already been discussed in chapter 4.6. No studies evaluating this scenario in the region or a comparable one could be identified. Notwithstanding, and as previously explained, the urease inhibitor is assumed to not affect the nitrogen supplied by the fixation of the legumes. Moreover, Espindula et al. (2014) assessed wheat yields in Brazil under varying numbers of urea applications and the interaction with the urease inhibitor. Their results indicate that yields are not significantly affected by the splitting of nitrogen applications in combination with the urease inhibitor when the total fertilizer rate is maintained. Furthermore, Shigueru Okumura et al. (2013) reported comparable corn yields in the state of Paraná fertilizing with urea treated with NBPT at rates similar to those assumed in this combination of strategies. Theoretically, this partially validates the feasibility of maintaining yields in the Typical Farm when the nitrogen rate is reduced to the rates assumed. Consequently, based on the studies presented, it is assumed that corn yields are not affected by the combination of the three mitigation strategies.

Total costs from implementing the mitigation strategy

The changes in the cost from optimizing the nitrogen rate, utilizing urea treated with urease inhibitors and using the cover crop mixture are presented in Table 5.12.

The mixture of cover crops increases the expenditure in the category “seeds” by 32 USD/ha. This cost represents the net difference between the seed costs in the baseline scenario and adopting the vetch–oat mixture. The fertilizer rate, which includes the treated urea, is reduced depending on the case assumed. The costs are reduced in the three cases: 32 USD/ha in the normal case, 46 USD/ha in the best case and 17 USD/ha in the worst case.

The additional insecticide spraying for the cover crops increases the cost of pesticides by 12 USD/ha. Diesel and labor both increase due to the additional farm operation, totaling an increase of 2 USD/ha. Machinery costs increase by 0.3 USD/ha from the additional repairs and maintenance from the insecticide application, however, this value is not presented as it rounded down to zero. Contractor costs are increased by 10 USD/ha, as this service is assumed for the additional pass to spread urea. The cost changes in these categories are the same for the three cases.

Liming is included in the category “remaining,” as it is not affected by the combination of strategies.

The total cost of the combination of strategies in corn is 23 USD/ha in the standard case, 9 USD/ha in the best case and 39 USD/ha in the worst case. These are used to calculate the mitigation cost.

Table 5.12 Corn costs with combination of strategies for Typical Farm in Paraná (USD/ha)

	Baseline scenario	Standard case		Best case		Worst case	
		Strategy	Change	Strategy	Change	Strategy	Change
Seed	261	293	32	293	32	293	32
Fertilizer	393	361	-32	347	-46	376	-17
Pesticides	163	174	12	174	12	174	12
Diesel	19	20	1	20	1	20	1
Labor	10	11	1	11	1	11	1
Contractor	0	10	10	10	10	10	10
Remaining	34	34	0	34	0	34	0
Total	**880**	**903**	**23**	**889**	**9**	**919**	**39**

Source: own estimation.

Emissions per hectare from implementing the mitigation strategy

The reduction of GHG emissions in corn resulting from the implementation of the combination of strategies is presented in Table 5.13. The emissions from direct decrease by 235 kg CO_{2eq}/ha in the standard case, 353 kg CO_{2eq}/ha in the best case and 108 kg CO_{2eq}/ha in the worst case. In the case of indirect, emissions are lowered by 98 kg CO_{2eq}/ha, 155 kg CO_{2eq}/ha and 33 kg CO_{2eq}/ha in the respective cases. The changes in both categories are largely attributed to the reduction in the nitrogen fertilizer used, which drives the emissions of N_2O. The use of vetch in the cover crop mixture adds more nitrogen to the crop residues that can be lost as N_2O, increasing GHG emissions, yet this effect is comparatively small.

Emissions from urea also decrease because of the adjusted fertilizer rate, resulting in a reduction of 84 kg CO_{2eq}/ha in the standard case, 109 kg CO_{2eq}/ha in the best case and 57 kg CO_{2eq}/ha in the worst case. Correspondingly, the emissions from the production of the fertilizer, shown in the category "manufacture," are lowered by 116 kg CO_{2eq}/ha in the standard case, 150 kg CO_{2eq}/ha in the best case and 79 kg CO_{2eq}/ha in the worst case.

Diesel emissions increase by 7 kg CO_{2eq}/ha in all cases, which results from the additional passes to spread fertilizer and apply insecticide. The categories liming and land use are not affected by the strategies and hence are included in remaining.

The total change in GHG emissions from corn in the Typical Farm in Paraná is a reduction of 526 kg CO_{2eq}/ha in the standard case, 760 kg CO_{2eq}/ha in the best case and 270 kg CO_{2eq}/ha in the worst case. These reductions are used to calculate the mitigation cost per ton of CO_2.

Table 5.13 GHG emissions in corn with combination of strategies for Typical Farm in Paraná (kg CO_{2eq}/ha)

	Baseline scenario	Standard case		Best case		Worst case	
		Strategy	Change	Strategy	Change	Strategy	Change
Direct	1,957	1,722	-235	1,604	-353	1,849	-108
Indirect	226	128	-98	71	-155	193	-33
Urea	301	216	-84	191	-109	244	-57
Manufacture	602	486	-116	452	-150	524	-79
Diesel	63	70	7	70	7	70	7
Remaining	551	551	0	551	0	551	0
Total	**3,700**	**3174**	**-526**	**2,939**	**-760**	**3,430**	**-270**

Source: own estimation.

Mitigation costs and considerations on the adoption, monitoring and enforcement of the strategy

The mitigation cost from the combination of strategies is 44 USD/t CO_{2eq} in the standard case, 12 USD/t CO_{2eq} in the best case and 143 USD/t CO_{2eq} in the worst case.

Possibly, the implementation of the combination of strategies could be realized in the subsequent growing season as no investment in equipment or new know-how is required. A wide-scale adoption also should be feasible since no additional competition for the machinery at the farm level results from the joint implementation of the individual strategies.

No additional considerations regarding monitoring and enforcing of the strategy are assumed for the combination. The individual strategies would need to be monitored independently from each other. Thus, the assessments presented for the individual strategies are likewise valid without significant changes.

The adoption of the strategies could be promoted stepwise, as previously discussed in chapter 4.5.2. This type of approach would facilitate acceptance by the growers, as they could learn the intricacies of the strategies on a smaller acreage without incurring a higher risk. Moreover, an advantage of promoting the combination of strategies is the possibility to utilize one approach to encourage the implementation of more than one strategy. For instance, a tax on nitrogen fertilizer would improve the economics of the three strategies, as they all lower the input of fertilizer needed. Consequently, the total transaction cost of the combination could be lower than the sum of the costs of the individual strategies.

5.7 Policy advice: ranking of GHG mitigation strategies

The mitigation costs of all the strategies assessed for corn production in the Typical Farm in Paraná are presented in Table 5.14. As already conducted in chapter 4.7, the strategies are ranked based on their mitigation costs. However, since no strategy leads to carbon sequestration, no different time horizons are considered, as the strategies remain unchanged over time.

Table 5.14 Comparison of costs in the standard, best and worst cases for Typical Farm in Paraná (USD/t CO_{2eq})

Strategy	Code	Standard case	Best case	Worst case
Optimization of nitrogen rate	BR-1	-10	-26	206
Cover crops	BR-2	70	42	115
Enhanced efficiency fertilizer (inhibitors)	BR-3	134	103	155
Combination of strategies	BR-C	44	12	143

Source: own estimation.

Based on the standard case as a reference, the optimization of the nitrogen rate offers the lowest mitigation cost, which is -10 USD/t CO_{2eq}. This is the only strategy that results in a win-win scenario. The second most economic mitigation strategy is the combination, which implies a cost of 44 USD/t CO_{2eq}. The following strategy in the ranking is the cover crop mixture, which costs 70 USD/t CO_{2eq}. Finally, the most expensive approach are the enhanced-efficiency fertilizer, posing a cost of 134 USD/t CO_{2eq}. Thus, the use of these fertilizers is approximately three times costlier than the best strategy with positive costs (combination).

The ranking of strategies is the same when the best case is assumed. However, assuming the worst case, cover cropping becomes the most economical alternative. The combination of strategies is indicated to be the second-best alternative in economic terms. The adoption of urease inhibitors is the third-best option, and the optimization of the nitrogen rate becomes the costliest strategy.

5.8 Summary and main findings of the case study

Three different GHG mitigation strategies and their combination were assessed for Paraná Typical Farm:

(1) The nitrogen rate can be reduced by adding one more application of fertilizer. This additional pass is conducted by a contractor. The lowered fertilizer rate compensates for the costs of the contractor, except when the lowest reduction potential is assumed.

(2) The cereal cover crop used in the baseline can be replaced by a mixture of legume and cereal. No net change in the soil carbon content takes place, but the legume fixates nitrogen, which reduces the need to use synthetic fertilizers.

(3) Urease inhibitors allow a reduction of the urea rate. However, they imply a cost higher than the savings stemming from the lower rate. No changes or adjustments are required in the operations, but their reduction potential can be regarded as low.

(4) It is feasible to implement the combined strategies without making further changes. It combines the technical and agronomic changes entailed in each strategy. The total changes imply an increase in corn costs ranging from 1% to 4% In the best and worst cases. In the standard case the increase is approximately 2%. Nonetheless, the highest mitigation is achieved.

Debatably, the mitigation cost of the strategies varies greatly. Based on mitigation cost-efficiency, the optimization of the nitrogen rate ranks as the best alternative as it has a negative cost. From the strategies with positive costs, the combination of strategies is the optimal strategy.

6 Case study 3: Germany – winter wheat in Mecklenburg-Vorpommern

The criteria used to select Germany as the region within the EU for the case study are explained in chapter 3.1. The relevance of the country's agricultural market share at the international level as well as collaboration with researchers and other experts participating in *agri benchmark* were considered in this decision.

The exchange rate assumed is 0.88 Euros per USD, which represents the average annual rate for the 2016-2018 period (World Bank, 2021). All calculations are done in the local currency and are transformed into USD as the last step.

The composition of this chapter and considerations described in this case study follow the same approach already presented in chapter 4 for the Typical Farm in Iowa. Therefore, in the cases where repetitions occur or no adjustments are necessary, the concepts are not explained in detail, as these already have been discussed. To better understand this case study, first read the assessment conducted for the Typical Farm in Iowa.

6.1 Selection of the region

The assessment to determine the region for the case study is first looks at state-level statistics, as most information is provided at least at this level of disaggregation. In this regard, the output of winter wheat is used as an indicator of the relevance of the regions in the national context. According to DESTATIS (2018, 2020b), the states producing the largest shares of national wheat production (average 2017-2019) are Bayern (16.5%), Niedersachsen (13.1%) and Mecklenburg-Vorpommern (10.1%). Scientists in the three states were contacted to present this project and offer the possibility to cooperate.

Utilizing the *agri benchmark* network, collaboration with researchers in Mecklenburg-Vorpommern was gained, which is key for this assessment. Moreover, the scientific partners responsible for the preexisting Typical Farm in the region, used for the construction of the preliminary background for the case study, agreed to support the project. An additional consideration for this decision was a strong interest in the topic and in working on projects related to GHG mitigation in crop rotations in the region, meaning familiarity with the discussion and considerations related to this topic, which can support the project with technical expertise. Hence, considering the interest and knowledge of the scientific partner in this topic as well as his previous experience with the focus group approach and the concept of the Typical Farm, Mecklenburg-Vorpommern was selected for the case study.

6.2 Climate

The mean climate values in the region are shown in Table 6.1. The average annual precipitation is 640 mm, which is distributed across the year evenly. However, between March and June, it is possible to have comparatively long periods with low rainfall (LFA, 2014a). The risk of frost begins in late October and extends until late March; snow can fall from November until February (DWD, 2022).

Table 6.1 Climatic data in Mecklenburg-Vorpommern (1981-2010)

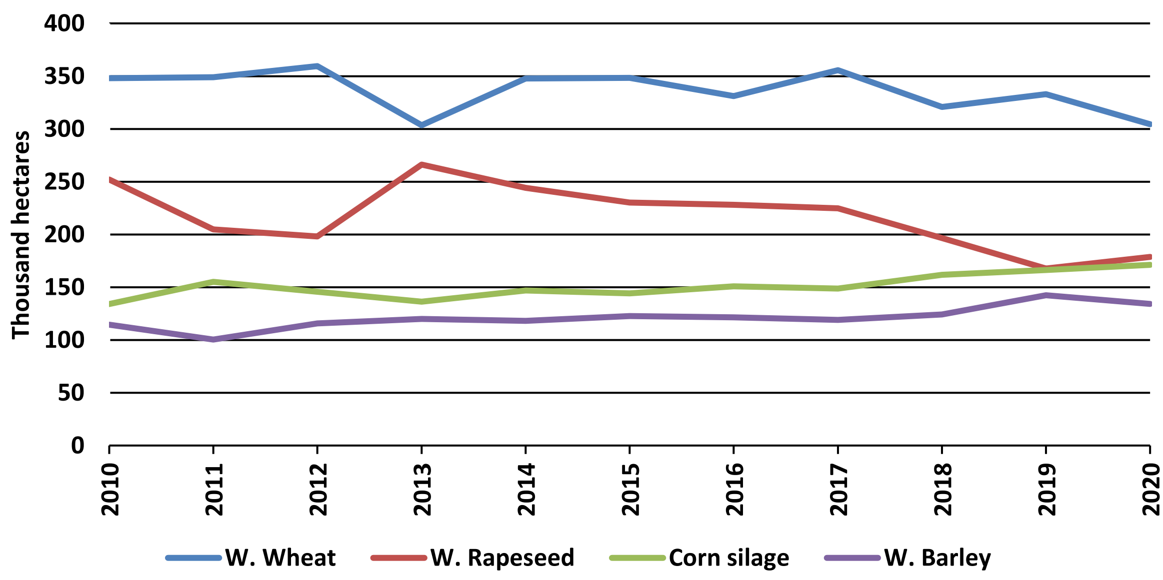
Month	Temperature (°C)	Min. Temp. (°C)	Max. Temp.(°C)	Monthly Rainfall (mm)
Jan	0.8	-8.7	5.3	54
Feb	1.3	-10.5	6.1	41
Mar	4.0	-1.3	7.3	49
Apr	8.3	4.3	12.0	39
May	8.9	8.6	16.7	53
Jun	15.5	11.7	20.0	61
Jul	18.1	13.9	22.5	70
Aug	17.6	13.6	21.7	63
Sep	13.8	9.8	17.5	52
Oct	9.5	5.0	12.7	51
Nov	4.9	0.0	7.8	52
Dec	1.5	-5.1	7.1	55

Source: DWD (2022)

6.3 Background and key features of arable farming in Mecklenburg-Vorpommern

There are approximately 1.1 million ha of arable land in Mecklenburg-Vorpommern (DESTATIS, 2022). Based on the average acreage from 2010 to 2020, the most-grown crops are winter wheat (31%), winter rapeseed (21%), corn silage (14%) and winter barley (11%). These crops combined represent 76% of the total acreage in the state. Usually, winter wheat is grown after winter rapeseed (Bull, 2018). The acreage of these crops is presented in Figure 6.1.

Figure 6.1 Acreage of selected crops in Mecklenburg-Vorpommern from 2010 to 2020



Source: own elaboration based on DESTATIS (2022)

The acreage of winter wheat has remained mainly constant since 2010. However, the production of winter rapeseed has a downward trend. This partly can be attributed to the intensity of nitrogen used in this crop and the implementation of the Fertilizer Directive or Düngerverordnung (DüV). The guidelines of the BMEL (2021a), which has the objective to restrict the input of nitrogen fertilizer to lower the risks of pollution, determine the nitrogen rates that can be applied. These rates are based on the crop grown, historical yields and mineral nitrogen available in the soil[30] from the previous season, among other factors. Rapeseed's low nitrogen uptake rate implies that high fertilizer rates may be required to maximize economic return (Bouchet et al., 2016), and with the DüV limits coming into force, this crop has become relatively less profitable. Therefore, growers have been gradually replacing this crop. Pressure from pests has further decreased the economic performance of this crop (LFA, 2019b).

Since 2020, the BMEL (2021a) makes the use of urease inhibitors mandatory when urea is applied and cannot be incorporated within four hours after it is spread. Thus, it can be inferred that all urea applications occurring after seeding already include these inhibitors, since incorporating the fertilizer when the crop is already established would damage the plants. Moreover, this implies that the mitigation strategy based on using enhanced-efficiency fertilizers already is partly included in the baseline scenario[31].

Growers in the EU receive economic support from the Common Agricultural Policy (CAP). To receive the basic payments, growers are required to comply with rules that define, among other things, the least number of crops in the rotation as well their maximum shares. These payments are decoupled, meaning that they are not tied to a specific crop.

6.4 Typical Farm and focus group

The preexisting Typical Farm was used for the preliminary baseline scenario in the discussion with the focus group. Adjustments to this baseline as well as changes required for the mitigation strategies follow the approach explained in chapter 3.4.1.

6.4.1 Typical Farm and its representativity

Following the practice in chapter 3.4, the Typical Farm is compared to survey data to assess its representativity. The Typical Farm is assumed to have 1,100 ha dedicated exclusively to the production of cash crops. The latest Agricultural Structure Survey (Agrarstrukturerhebung), conducted in 2016, provides data on the acreage operated by various farm size classes. Based on the survey's description, the Typical Farm is in the category of farms with 500 ha or more, which is the biggest of the size classes. Farms in this class manage 68% of the arable land of the state (DESTATIS, 2022). Fundamentally, this confirms that the Typical Farm belongs to the category producing the bulk of the output, as defined in the Typical Farm approach. However, considering the range of the farm sizes that may potentially be included in the survey's size class, this characteristic of the Typical Farm cannot be sufficiently evaluated.

The Typical Farm is assumed to have reduced tillage, which is used on approximately 60% of the acreage in Mecklenburg-Vorpommern (DESTATIS, 2010). The crop rotation assumed is winter rapeseed, winter wheat, corn silage and, lastly, a winter cereal. This cereal can be assumed to be barley and wheat, both with equal shares. No statistical data on crop rotations could be identified, although this type of crop rotation has been deemed common in Mecklenburg-Vorpommern in reports by LFA (2019b)

---

30 In the context of the values and literature concerning the DüV, the mineral nitrogen available in the soil is referred to as N_{min}.

31 The possibility to combine the urease inhibitor with other additives to enhance the mitigation effect was assessed. However, no applicable scientific literature could be identified. Hence, the possibility to combined two or more additives is not considered.

The input intensity and yield of winter wheat depend on the crop preceding it (LFA, 2019b). Considering that this crop is grown twice in the rotation and after different crops, a choice must be made. The share of the crop in the rotation is used as the determining criteria to select the crop, as this theoretically increases the representativity of the results. Thus, winter wheat preceded by winter rapeseed is used for this case study.

In addition, it can be assumed that the production strategy of this wheat in the Typical Farm is to fulfill the criteria to reach baking quality[32]. Baking wheat has a higher grain price than feed wheat, which is the category attained if the criteria are not met. The input intensity required to reach baking quality - particularly nitrogen usage to generate the protein content necessary (Martre et al., 2006) - is usually higher. The particularities of producing baking wheat are taken into account in this comparison and the case study.

Selected agronomic and economic indicators for Mecklenburg-Vorpommern collected from farms distributed across the state are made available by LFA. These indicators are used for comparison against the Typical Farm, which is shown in Table 6.2. The average from the reported data from 2018 to 2020 is used to depict a “normal” year.

The average wheat yield for the Typical Farm is 19% higher than in LFA’s reports, but the nitrogen rate and protein contents are similar in both cases. The protein content is one of the key criteria to determine baking quality in Germany (Vollmer and Mußhoff, 2018). Fertilizer expenditure on the Typical Farm is higher than the average from LFA. However, considering that no information on the specific type of fertilizers, rates or unitary prices is provided, it is not possible to make further inferences. Crop care is the category with the largest difference, with it significantly higher in the Typical Farm. However, given that no details are provided, no further analysis is possible. Arguably, most of these differences can be attributed to the agroclimatic variability observed within the state.

Table 6.2 Comparison of winter wheat for Typical Farm with data from LFA

	Typical Farm	Estimation LFA	Difference*
Yield (t/ha)	9.0	7.6	19%
N rate (kg/ha)	193.0	194.3	-1%
Protein content (%)	12.5	12.8	-2%
Fertilizer (USD/ha)	207.0	177.4	17%
Crop care (USD/ha)	227.4	173.2	31%

*Considers the estimation as the reference point to compare it with the Typical Farm.
Source: own estimation based on LFA (2019a, 2020, 2021).

While certain aspects of winter wheat production on the Typical Farm are in line with average state data (nitrogen rate and protein content), some indicators such as yields differ significantly. Consequently, despite winter wheat production on the Typical Farm having characteristics similar to the average of the state, the representativity of the results is provided by the focus group methodology.

6.4.2 The Mecklenburg-Vorpommern focus group

The Typical Farm already depicted in Mecklenburg-Vorpommern was used for the preliminary evaluation of the mitigation strategies. Expert interviews and scientific literature also were reviewed to prepare the

[32] Baking quality is divided in several subcategories’ dependent on the baking properties of the grain. This case study focuses on wheat of B quality, which was considered in the focus group discussion.

focus group discussion, which was conducted in October of 2021. Five growers, two researchers including the person responsible for the Typical Farm, an agricultural consultant who also participates in the *agri benchmark* network and the researcher conducting this assessment participated. The period 2018 to 2020 was used to define an average year in the focus group as well as for the comparison with statistical data. The input prices depict the situation before the coronavirus pandemic since it generated significant market disruptions.

The discussion started with an introduction of the project followed by the presentation of the preliminary results from the Iowa and Paraná case studies. The production of winter wheat in the baseline scenario was defined based on the discussion in the focus group and the preexisting Typical Farm in the *agri benchmark* network. The results and values assumed for the Typical Farm are presented in chapter 6.5.1.

After the baseline was established, the discussion of the GHG mitigation strategies was conducted. Each strategy was evaluated until a consensus was reached among the participants. Although the strategies focused on winter wheat production, the repercussions on the entire crop rotation were considered. Additional interviews with experts and a revision of previous studies were used to validate the strategies and their consequences.

6.5 Mitigation strategies analyzed

The approach used in this case study has been explained in chapter 4.5 for the Typical Farm in Iowa. Hence, the relevant features of the mitigation strategies already covered in that case study are not discussed in detail again. Only the key aspects, as well as the differences applicable to this Typical Farm, are explained in this assessment.

The possibility to combine the mitigation strategies assessed for the Typical Farm is presented in chapter 6.6 and the comparison of the strategies in chapter 6.7. Appendix A contains the tables presenting the assumptions used for the calculation of the mitigation costs in this case study[33].

6.5.1 Status quo – baseline scenario (DE-0)

The baseline scenario derived from the adjustments to the preexisting Typical Farm and additions from the focus group are discussed in this chapter. The GHG mitigation strategies presented are compared against this baseline. The total expenditure in winter wheat production is 828 USD/ha, which is disaggregated in Table 6.5 in chapter 6.5.1.

The data on yields, protein content, fertilizers, pesticides, labor and diesel are obtained from the focus group. However, the machinery costs (depreciation and repairs) assumed in the baseline scenario are adopted from the preexisting Typical Farm in the *agri benchmark* network. Thus, these machinery costs do not include the adjustments made in the baseline scenario generated in the focus group discussion. Nonetheless, presenting the machinery costs from the Typical Farm in the network is considered valuable information to understand how winter wheat is produced in Mecklenburg-Vorpommern. It must be noted that adopting this value does not affect the calculation of the mitigation costs in the case study, as only the marginal change necessary for the implementation of the strategies is needed.

[33] Input costs are presented in Table A.1, coefficients for the GHG calculation in Table A.2, inputs used in wheat in Table A.10, description of machinery in Table A.12, and farm operations in Table A.13.

Emissions per hectare in the baseline scenario

For the estimation of the adjustment factor for indirect emissions (see chapter 3.5.1.2) and total GHG emissions from winter wheat production, nitrogen fertilizers must be evaluated based on their types. In this Typical Farm, the total fertilizer rate (193 kg N/ha) is applied as urea treated with urease inhibitors[34] (120 kg N/ha), diammonium phosphate (or DAP) (18 kg N/ha) and urea ammonium nitrate (or UAN) (55 kg N/ha).

The DüV (BMEL, 2021b) makes it mandatory for growers to sample the soil to report the amount of nitrogen from the previous crop that remains in the soil and is available for the following crop. In this regard, the participants of the focus group mentioned that, on average, 33 kg N/ha are available in the soil for winter wheat after rapeseed. In turn, the nitrogen after winter wheat is reported to be 23 kg N/ha. This implies a net reduction of 10 kg N/ha in the nitrogen reserves of the soil occurring during the wheat production season. This nitrogen can be assumed to exit the system in the harvested crop parts and be lost to the environment via direct and indirect pathways. Consequently, for the calculation of the mitigation strategies and emissions[35], the 10 kg N/ha are deemed as an additional source of nitrogen not included in the fertilizer rate. Hence, the emissions allocated to winter wheat are higher than just considering the nitrogen fertilizer applied directly to this crop.

The moisture content of the wheat is assumed to be 14%, which is the common value used for this crop in Germany. The nitrogen content of the grain obtained with 12.5% protein content is assumed to be 1.89% of the fresh matter (or 2.33% of the dry matter), based on the reference values from the BMEL (2021a)[36]. Consequently, considering a yield of 9 t/ha, a total of 170 kg N/ha are removed from the system via harvest.

The climate of the Typical Farm according to the IPCC categories is defined as Cool Temperate Moist (JRC, 2022). The GHG emissions from winter wheat production in the Typical Farm in Mecklenburg-Vorpommern are 3,007 kg CO_{2eq}/ha and are presented in Table 6.3. A summary of the emissions entailed in each category is provided in chapter 3.5.7.

---

34 For the calculation of GHG emissions from urea treated with urease inhibitors, the default emission factors from IPCC (2019a) are adopted. The rate of this fertilizer is constant in all strategies and cases. Consequently, this assumption does not affect the mitigation costs, as only the changes between the emissions in the baseline scenario and the mitigation strategies are required.

35 The result of the soil sampling according to the BMEL (2021a) represents the sum of the nitrogen in the soil available as NH_3 and as NO_3. For the calculation of the emissions according to IPCC (2019a), the 10 kg N/ha are assumed to be 50% NH_3 and 50% NO_3. No costs or emissions from the manufacture of this nitrogen are assumed.

36 The reference values provided by BMEL (2021a) do not include winter wheat with 12.5% protein content. Hence, this value is obtained utilizing linear interpolation from the values indicated.

Table 6.3 GHG emissions from winter wheat for Typical Farm in Mecklenburg-Vorpommern

	Hectare ($kg\ CO_{2eq}/ha$)	Product ($kg\ CO_{2eq}/t$ crop)
Direct	1,858	206
Indirect	162	18
Liming	0	0
Urea	235	26
Manufacture	471	52
Diesel	282	31
Land use	0	0
Total	**3,007**	**334**

Source: own estimation.

No lime is applied to winter wheat. Moreover, it can be assumed that the Typical Farm has not changed the tillage regime or the input of biomass in the past 20 years. Therefore, according to IPCC (2019a) methodology, no net change in the carbon content of the soil takes place (see chapter 3.5.6); thus, no emissions from land use are calculated.

6.5.2 Optimization of nitrogen rate (DE-1)

As explained in chapter 3.6.2.1, it theoretically is possible to mitigate emissions by reducing the nitrogen rate to lower the losses without affecting the yield. The NUE approach can be used to determine the nitrogen reduction potential. The total nitrogen supply considered for this calculation is assumed to be the sum of the fertilizer rate (193 kg N/ha) as well as the difference in the nitrogen reserves in the soil (10 kg N/ha). The export of nitrogen as harvest is calculated to be 170 kg N/ha; thus, the Typical Farm has an NUE of 84% in the baseline scenario, which can be regarded as high.

The default reference values presented in chapter 3.6.2.1 are used for the optimal NUE, as no region-specific estimates could be identified in scientific literature. These optimal values are 80% for the worst case, 85% for the standard case and 90% for the best case. However, the NUE in the baseline already is higher than the optimal value used in the worst case (84% and 80%). This implies that implementing the optimal NUE indicated by the worst case (80%) results in a higher fertilizer rate (202 kg N/ha) than the one used in the baseline scenario, which would generate comparatively more emissions. Hence, the worst case is omitted in this strategy. The NUE from the standard and best cases are adopted without adjustments. The fertilizer rates and nitrogen reduction are presented in Table 6.4.

Table 6.4 Reduction of nitrogen rates based on optimal NUE for Typical Farm in Mecklenburg-Vorpommern

Case	N removed via harvest (kg N/ha)	Fertilization rate (kg N/ha)	NUE (%)	N reduction potential (kg N/ha)
Baseline	170	193	84	-
Standard	170	190	85	3
Best case	170	179	90	15

Source: own estimation based on Brentrup and Palliere (2010).

The participants of the focus group explained that the nitrogen fertilization program used in winter wheat begins with an application of urea as well as DAP in February, followed by UAN in April and, lastly, UAN in May. The rate applied with UAN each time is 28 kg N/ha, which is mixed with pesticides. Considering the number of passes already included in the baseline scenario and the competition for machinery stemming from the other crops, the growers commented that adding more passes is not feasible. Still, they added that the UAN rate could be decreased by an equal amount in each of the two passes. This implies that no changes in machinery, diesel or labor are necessary, as the existing fertilization program is used.

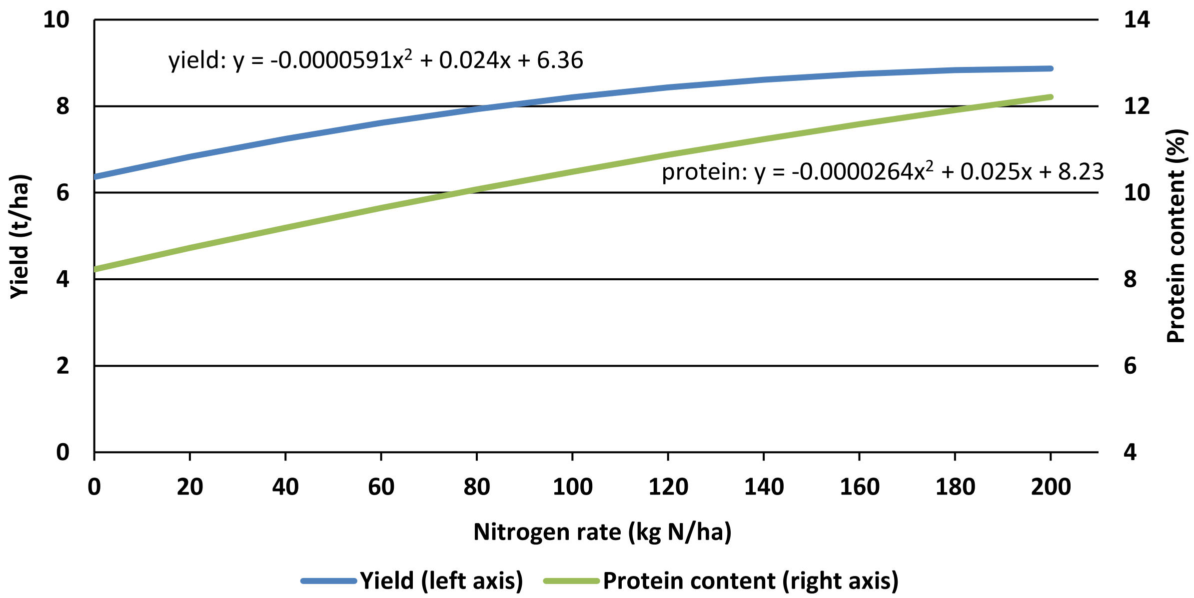
However, they also mentioned that to minimize the risk of negatively affecting the wheat yield, they would hire a contractor to fly a drone with nitrogen sensors before the last application to detect possible deficiencies and compensate if it is necessary. The contractor fee is assumed to be 28 USD/ha.

The participants of the focus group stated they are aware that they operate on the higher levels of wheat's nitrogen-response curve. They added that they estimate the marginal yield change from the reduced nitrogen rates to be small, but no specific value was provided. Moreover, they mentioned that the reduced rate could affect the protein content, decreasing the quality of the produce, resulting in a decreased price received for the wheat.

To evaluate these potential variations, the results published by Michel et al. (2007), conducted in the same region as the Typical Farm, are used as a reference. Between 1993 and 2000, they tested several wheat varieties based on the baking quality. Based on the focus group, the wheat produced in the Typical Farm is assumed to be B-quality. Two different sites with this wheat quality are reported in the analysis, representing the usual agroclimatic conditions found in the region. For each site, a nitrogen yield response curve is provided. Thus, an average of both curves is calculated and used to quantify the possible changes of the nitrogen reduction on the yields in the Typical Farm.

Moreover, Michel et al. (2007) also provide a regression for the estimation of the protein content as a function of nitrogen fertilizer, although for only one of the sites. The other regression is omitted as it did not fit the statistical presumptions. Thus, the effect of the nitrogen reduction on the protein content is based on only one regression. The yield and protein curves are shown in Figure 6.2.

Figure 6.2 Estimated wheat yield and protein content under different nitrogen fertilization rates in Mecklenburg-Vorpommern



Source: own estimation based on Michel et al. (2007).

Calculating the regression using the nitrogen rate in the baseline scenario results in an estimated yield of 8.9 t/ha. Taking into account that the yield of the Typical Farm is 9 t/ha, the regression is deemed an adequate representation. Compared with the yield calculated with the regression, the reduced nitrogen rates cause a decrease of 0.01 t/ha in the standard case and 0.03 t/ha in the best case. These decreases can be considered negligible.

The same approach is used to assess the effect of nitrogen reduction on protein content. The curve is also deemed a good approximation, as it results in 12.1% protein content. The highest nitrogen reduction (best case) lowers the calculated protein content by 0.2%. Deducting this difference from the protein content in the baseline scenario (12.5%) results in 12.3%. Usually, a protein content of at least 12% is required to reach the B-quality in Mecklenburg-Vorpommern (Vollmer and Mußhoff, 2018). Assuming the protein content is affected by the nitrogen reduction assumed in the mitigation strategies, the wheat would still attain the same quality and, thus, its grain price.

Possibly, reducing nitrogen fertilizer could affect the nitrogen available after winter wheat. The potential effect that the nitrogen reduction could have depends on multiple variables, including crop, fertilizer type and timing of the application, among other factors. No reference values for the region or an applicable approach to calculate this change could be identified. However, the results from LfL (2001) can be used as an indication of the effect. They measured the nitrogen available after several crops for 10 years in 214 different sites distributed in the state of Bayern. The authors found that crops fertilized with less than 100 kg N/ha and with 100 to 200 kg N/ha left approximately the same amount of nitrogen in the soil. A likely explanation for this phenomenon is that the crops had not reached their yield potential and were therefore absorbing all the nitrogen that was easily available to them. Considering that the nitrogen rates calculated for the Typical Farm are within the ranges mentioned in the study, the nitrogen available in the soil after winter wheat is assumed to remain unchanged by the strategy.

Total costs from implementing the mitigation strategy

The total cost of optimizing the nitrogen fertilization rate of winter wheat in the Typical Farm is shown in Table 6.5. The worst case is omitted as the NUE in the baseline scenario is higher than the optimal value assumed for this case.

The reduction in fertilizer expenditure varies depending on the case, as the UAN reduction achievable differs. The standard case results in a reduction of 3 USD/ha and the best case in a reduction of 11 USD/ha. The contractor's fee to fly the drone to measure the nitrogen levels is 28 USD/ha, which is the same in both cases. No further changes result from the implementation of this mitigation strategy. Thus, the total change in the standard case is an increase of 26 USD/ha. The change in the best case is an increase of 18 USD/ha. Thus, even the highest UAN reduction is not sufficient to compensate for the added cost of the contractor. These values are used for the calculation of the mitigation cost.

Table 6.5 Wheat costs with optimized nitrogen fertilization rate in Typical Farm in Mecklenburg-Vorpommern (USD/ha)

	Baseline scenario	Standard case		Best case	
		Strategy	Change	Strategy	Change
Seed	60	60	0	60	0
Fertilizer	207	204	-3	196	-11
Pesticides	227	227	0	227	0
Liming	0	0	0	0	0
Diesel	87	87	0	87	0
Labor	80	80	0	80	0
Machinery	167	167	0	167	0
Contractor	0	28	28	28	28
Total	**828**	**854**	**26**	**846**	**18**

Source: own estimation.

Emissions per hectare from implementing the mitigation strategy

The reduction in GHG emissions derived from the reduced nitrogen rate is presented in Table 6.6. The emissions from the categories direct and indirect, both mainly driven by the input of nitrogen, are lowered depending on the case. In the standard case, emissions from direct are reduced by 26 kg CO_{2eq}/ha and from indirect by 18 kg CO_{2eq}/ha. The reductions in the best case are 109 and 79 kg CO_{2eq}/ha, respectively.

Similarly, the reduction in the UAN rate lowers CO_2 emissions derived from the chemical structure of the fertilizer by 3 kg CO_{2eq}/ha in the standard case and by 12 kg CO_{2eq}/ha in the best case. The emissions from the manufacture of fertilizer are decreased depending on the fertilizer reduction calculated. Here, a decrease of 9 kg CO_{2eq}/ha is calculated for the standard case and 40 kg CO_{2eq}/ha in the best case. No further changes are estimated to occur; hence, the categories liming, diesel and land use are included in remaining.

The resulting emissions from the standard case are a net reduction of 56 and of 239 kg CO_{2eq}/ha in the best case. These values are used to calculate the mitigation cost from the strategy.

Table 6.6 GHG emissions from winter wheat with optimized nitrogen fertilization rate for Typical Farm in Mecklenburg-Vorpommern (kg CO_{2eq}/ha)

	Baseline scenario	Standard case		Best case	
		Strategy	Change	Strategy	Change
Direct	1,858	1,832	-26	1,749	-109
Indirect	162	143	-18	83	-79
Urea	235	232	-3	224	-12
Manufacture	471	461	-9	430	-40
Remaining	282	282	0	282	0
Total	**3,007**	**2,951**	**-56**	**2,768**	**-239**

Source: own estimation.

Mitigation costs and considerations on the adoption, monitoring and enforcement of the strategy

Reducing the nitrogen rate in winter wheat on the Typical Farm results in a mitigation cost in the standard case of 460 USD/t CO_{2eq} and of 73 USD/t CO_{2eq} in the best case.

The essential elements of monitoring this type of mitigation strategy as well as its enforcement already were addressed in the Iowa case study in chapter 4.5.2. However, an important difference results from Germany's Fertilizer Directive presented in BMEL (2021a). Under the current directive, growers are obliged to report their application of nitrogen, providing a detailed description of rates, fertilizer used and crop, among other factors. This information is reported to the state annually. Hence, the technical and organizational systems already have been implemented for this regulation. The same systems could be expanded or adjusted to better monitor the adoption of the mitigation strategy. Hence, compared with Iowa, which lacks this type of system, the transaction costs likely would be comparatively lower or potentially negligible in Mecklenburg-Vorpommern, as the current system would require only partial adjustment.

Comparison of results from Typical Farm with literature

Karatay et al. (2019) simulated the effect of a nitrogen reduction in winter wheat production in the state of Brandenburg, the southern neighboring state of Mecklenburg-Vorpommern. The simulations use a similar initial nitrogen rate as well as reductions compared to the assumptions for the Typical Farm. They estimate a reduction of 99 kg CO_{2eq}/ha, which is between the standard and best cases in this thesis (56 and 239 kg CO_{2eq}/ha). The mitigation costs they present are -111 USD/t CO_{2eq}, indicating that it is a win-win scenario. This differs significantly from the Typical Farm, as the costs are estimated to range from 73 to 460 USD/t CO_{2eq} in the best and standard cases, respectively. The considerable difference can be attributed to the different assumptions regarding input costs, especially the usage of the drone in the Typical Farm, which is not considered by the authors of the other study.

6.5.3 Cover crops (DE-2)

Change in the CAP regulations concerning cover crops

An important consideration regarding the assessment of this strategy is the likely change in the new CAP valid from 2023. When the case study was organized, the CAP (valid until 2022) considered the use of cover crops as a voluntary practice that could be subsidized if certain criteria were met (types of crops, number of species, etc.). However, in the proposal for the new CAP from 26/11/2021 (BMEL, 2021b), which was the newest version while assessing the strategy, it was stipulated that arable land should be covered from December the 1st to the 15th of January to lower the risk of erosion. The possible covers accepted in the proposal are perennial crops, winter crops, cover crops or not tilling the stubble from the previous crop (BMEL, 2021b). No specific definition in terms of the number of species or composition of the cover crop is stipulated in the proposal. Having a cover is mandatory to receive the basic payment of the CAP. Considering the relevance of this subsidy to crop production, it can be assumed that the Typical Farm will fulfill this requirement, indicating that this strategy will be implemented by the Typical Farm. It must be noted that no critical differences affecting the assumptions utilized in this case study were identified between the proposal of the CAP used to assess the strategy and the version of the CAP that was approved.

Assessment of the mitigation strategy

Approximately 33% of the arable acreage in Mecklenburg-Vorpommern was cultivated with summer crops in 2020 (DESTATIS, 2022). In the same year, 12% of the arable acreage had winter cover crops (DESTATIS, 2020a). Thus, it can be inferred that 64% of the acreage before summer crops is fallow. In this regard, the

Typical Farm is no exception, as it does not use winter cover crops. These theoretically could be seeded after the harvest of winter wheat and before the summer crop corn silage. This instance is the only case in which cover crops could be adopted in the current four-year rotation, as the remaining three seasons have winter crops. In principle, this implies that carbon can be sequestered in the soil via incorporation of the additional biomass, which can be considered a GHG mitigation strategy (see chapter 3.6.2.3).

In theory, replacing winter wheat with a combination of winter cover crops followed by spring wheat could promote carbon sequestration. However, this was regarded as economically unfeasible by the participants of the focus group, who commented that they estimated yields would be halved. This is in line with statistical data from the state as it indicates that the average yield of spring wheat is approximately 50% lower than the winter strains (DESTATIS, 2020b). Moreover, the members of the focus group estimated that winter wheat produces more biomass than the sum of a winter cover crop and spring wheat.

A brief literature review supports this hypothesis. Suppose a spring wheat yield on the Typical Farm of 4.5 t/ha (50% of the winter wheat yield). Assuming 14% moisture in the grain, the yield's dry mass weighs 3.9 t/ha. The ratio of above-ground residue to yield, both as dry mass, can be assumed to be 1.3 (IPCC, 2019a). Thus, the above-ground residues would decrease by 5 t/ha. Considering a ratio of above-ground biomass (sum of yield and residues) to below-ground dry biomass of 0.23 (IPCC, 2019a), the below-ground residues would decrease by 1.2 t/ha. Therefore, the net change in residues from producing spring wheat instead of winter wheat is a reduction of 6.2 t/ha.

Trials by LFA (2014b) with winter cover crops conducted from 2010 to 2013 in the region of the Typical Farm determined that the highest output of biomass is achieved with white mustard, which produces 2 t/ha (as dry mass). Assuming an above-ground to below-ground ratio of 0.22 (IPCC, 2019a), approximately 0.4 t/ha of residues are found below the soil. Thus, a total of 2.4 t/ha of residues are produced with the highest biomass-yielding winter cover crop tested. This is not enough to compensate for the reduction in the residues from growing spring wheat instead of winter wheat. Consequently, the carbon content of the soil would be reduced from the reduced input of biomass, which means that CO_2 would be released. Thus, this alternative is not a feasible mitigation strategy.

A list of possible winter cover crops was discussed in the focus group and was evaluated based on their cost, limitations regarding diseases shared with cash crops and proneness to winter-kill. A high probability of winter-kill was deemed critical by the focus group, since this would ensure the crop is terminated, lowering the risk of affecting the establishment of the following crop. They added that the use of herbicides is becoming more limited due to changes in the regulations; e.g., general prohibition of spraying glyphosate beginning in 2024. Based on these considerations, oats were considered the cover crop that best fit the criteria. Hence, this cover crop is assumed to this assessment.

The participants of the focus group mentioned that the common seeding rate for cover crop oats is 80 kg/ha. This crop already is produced in the region; hence, farm-saved seed can be purchased at 0.23 USD/kg. Seeding can be assumed to take place in early September utilizing the same pass as stubble tillage. In the baseline scenario, this operation is conducted with a cultivator using the grower's equipment, on which a seed drill can be mounted. Thus, cover crop seeds can be drilled in the soil as the stubble is tilled. Hence, no additional diesel or labor are required for seeding. The cultivator is not affected by this adjustment, so no changes in depreciation costs are assumed as the same farm operations are conducted as in the baseline.

However, the seed drill is considered supplementary equipment not included in the baseline, which implies additional costs from depreciation and maintenance. This seed drill is assumed to cost 8,530 USD and requires yearly maintenance of 227 USD. Usually, the cultivator would be purchased together with the seed drill and, equally, they would be resold together. Therefore, the same (10-year) depreciation period as the cultivator is assumed for the seed drill. The annual utilization of the cultivator in the baseline scenario is assumed to be at approximately full capacity. This determines the residual value, which is 25% of the purchase value in these conditions. Taking into account that both pieces of equipment would be sold together, the same percentage is assumed for the residual value of the seed drill, despite it having a lower utilization as it is not mounted every time the cultivator is utilized. All the machinery-related costs from the seed drill (3.8 USD/ha) are allocated to winter wheat for this estimation.

Oats are prone to winter-kill in Mecklenburg-Vorpommern. Thus, the cover crop can be assumed to be terminated due to winter temperatures. The members of the focus group mentioned that two passes with the disk harrow would be used to cut the residues and incorporate them into the soil. These passes are additional to the operations conducted in the baseline scenario.

Yet, the members of the focus group added that the ban on glyphosate, expected to start in 2024, would imply changes to corn silage production. Currently, this herbicide is sprayed in March before corn is seeded. After the ban is in place, they plan to conduct a pass with the disk harrow to control weeds. This operation can be interpreted as the replacement for the herbicide application. Thus, it can be inferred that the long-term baseline scenario already includes a pass with the disk harrow. Consequently, the incorporation of the residues of the cover crops implies only one additional pass with the disk harrow.

This pass is assumed to take place in late February and does not compete with any other operation. For this reason, as well as considering the type of equipment, no additional depreciation is assumed for the disk harrow. However, an increased expenditure from additional repairs and maintenance for the equipment and tractor is assumed (1.7 USD/ha), as well as more diesel and labor required for the pass. These cost changes are allocated to winter wheat too.

For the calculation of emissions from the decomposition of crop residues, based on the assessment by LFA (2014b) already presented, the above-ground dry biomass of oats is assumed to be 1.5 t/ha[37]. The nitrogen in the oats is added to that of the residues from winter wheat.

Komainda et al. (2018) tested the effect of several cereal winter cover crops before corn silage in trials conducted for two years in northern Germany. The authors concluded that the cover crops did not affect silage yields. The report from LFA (2014b) previously mentioned does not report the yields of the crop grown after the cover crop. Nonetheless, it is mentioned that no distinguishable differences could be perceived in the early growth stages of corn silage compared to the crop grown after winter fallow. Thus, negative yield effects are not expected in this case, as indicated by the authors. No further literature from the region of the Typical Farm could be identified. However, based on data from similar agroclimatic conditions, it is possible to obtain further insights. Hunter et al. (2019) evaluated oat's influence in silage yield from 2012 to 2015 in Pennsylvania and came to the same conclusion regarding yield changes. Moreover, Hashemi et al. (2013) concluded after a three-year trial in Massachusetts that oats as cover crop could have a positive effect in situations of limited nitrogen availability. They attribute this effect to the capacity of the cover crop to absorb nitrogen and its proneness to winter-kill.

Generally, cover crops absorb a share of the nitrogen remaining in the soil that otherwise would be leached. This nitrogen is then released when the crop residues decompose, and the nitrogen mineralizes. Considering that oats are prone to die naturally in winter, the nitrogen they absorbed quickly becomes available to the following crop.

---

[37] The default values from IPCC (2019a) are used to determine the nitrogen addition from crop residues, which are presented in Table A.5 in the Appendix. Based on them and the trial results, the above-ground residues are assumed to add 11 kg N/ha and the below-ground residues 3 kg N/ha. Thus, the total of 14 kg N/ha are added to the crop residues from winter wheat.

In the case of the Typical Farm, the aforementioned LFA (2014b) study can be used as a reference to determine the potential changes in soil nitrogen from using cover crops. The authors report the nitrogen available in the soil in March and April, before corn is seeded. The nitrogen available in the soil in March, without cover crops, is 28 kg/ha and in April, 47 kg/ha. The values under winter-killed cover crops, which includes oats, are 35 kg/ha and 72 kg N/ha, respectively. The increase in April in both cases can be attributed to the mineralization of the nitrogen in the residues. The average of both months is assumed, which implies 38 kg N/ha without cover crops and 54 kg N/ha with cover crops. Thus, the difference of 16 kg N/ha is assumed to represent the nitrogen that was prevented from leaching by the oats in the Typical Farm and is then available to the subsequent corn crop.

To calculate the reduction in emissions and costs from the prevented leaching, it is assumed that the saved nitrogen is reduced from the first application in corn silage, which is as urea. This implies a GHG emission[38] reduction of 211.5 kg CO_{2eq}/ha and lowered expenditure of 11.1 USD/ha. These changes are accounted for in the cost and emission tables of winter wheat.

No scientific literature could be identified that evaluated the long-term effect in the soil organic carbon resulting from cover crops in the region of the Typical Farm. However, the data gathered for the German Agricultural Soil Inventory, which sampled over 3,000 sites representative of the common practices in Germany, reveals that cover crops promote the increase in the soil organic carbon in arable systems (Jacobs et al., 2020). The meta-analysis by Poeplau and Don (2015), already presented in chapter 4.5.4, evaluates the effect of cover crops on soil organic carbon across several climatic and soil conditions, including sites in Germany. The analysis also reveals a positive relationship between cover crops and carbon sequestration. While these studies are not specific to the region of the Typical Farm, they indicate that carbon sequestration likely would occur.

The methodology provided by IPCC (2019a) is used for the estimation of the carbon sequestration potential (see chapter 3.5.6). This methodology requires the definition of the soil category, which for the Typical Farm can be assumed to be Mollisol (JRC, 2022). The soil category dictates the base soil organic carbon content. As already applied in chapter 4.5.4 for the Iowa case study, IPCC's approach also provides coefficients representing soil management and input of organic matter, among other factors. These coefficients are multiplied with the base soil organic content to calculate the content under the specific conditions assumed.

The use of cover crops is included in the coefficient of input of organic matter, which is provided with a default coefficient as well as a range. Thus, the default coefficient is assumed to represent the carbon content in the standard case. The highest coefficient indicated in the range implies the highest soil carbon and is thus used to determine the best case. Equivalently, the worst case is calculated using the lowest coefficient.

As already discussed in chapter 4.5.4, the calculation for the carbon content in the soil based on the IPCC (2019a) methodology implies that cover crops do not promote carbon sequestration if the worst case is assumed. This results from the factor used to calculate the net change being the same in the worst case as in a scenario without cover crops. Nevertheless, based on the literature revised in the aforementioned chapter, the probability of cover crops not promoting sequestration is considered highly unlikely.

Following the IPCC (2019a), it is assumed that the potential to sequester carbon is finite. Once the soil reaches a new equilibrium, the potential can be considered realized and no net change occurs. Consequently, the assessment of the mitigation strategy is divided into a transition stage representing the period with carbon sequestration and a stable stage, which represents the new equilibrium. To estimate

[38] To calculate the total reduction in emissions from the reduction of the 16 kg N/ha, the default emission factors from IPCC (2019a) are used, which are presented in Table A.6 in the Appendix. However, no adjustment factor is calculated (see chapter 3.5.1.2), as the information for corn silage was not discussed in detail in the focus group.

the sequestration potential in the transition stage, the difference between the soil organic carbon in the baseline and with cover crops is calculated. This result is divided by 20, which represents the period IPCC (2019a) indicates is needed for the soil to reach the new equilibrium. Hence, the division depicts the annual change in carbon in the soil in the transition stage (see chapter 3.5.6).

Nevertheless, winter cover crops are seeded only before the summer crop, which is one of four crops in the rotation, the rest being winter crops. The IPCC (2019a) methodology does not provide data on how to calculate changes in the carbon content of the soil when only a share of the rotation can be modified to include cover crops. Hence, further assumptions are needed in this case. It can be inferred that the additional biomass amounting to carbon sequestration is generated only in the year with the cover crop. In the years without cover crops, the carbon content of the soil can be assumed to remain as in the baseline scenario as no additional biomass is generated. Thus, a share of the carbon sequestered in the year with the cover crops would be lost in the years without them, because the soil would receive comparatively less biomass. Following IPCC's methodology, a linear change of the carbon in the soil can be assumed. Hence, if all else is held constant, the carbon sequestration potential with cover crops in the Typical Farm can be inferred to be 25%, which represents the share of years in which the strategy is applicable. This assumption is applied to the net carbon sequestration potential calculated with the formula in chapter 3.6.2.3, indicating that the result from the formula is divided by four. It is recognized that this consideration relies on a simplification of the soil carbon dynamics. However, in absence of crop and region-specific values, it is deemed a proper approximation for this Typical Farm. The resulting annual GHG mitigation potential is presented in Table 6.7. The coefficients used from IPCC (2019a) for the calculation of the carbon contents are shown in Table A.15 in the Appendix.

Table 6.7 Soil organic carbon stocks and sequestration potential with cover crops over 20 years for Typical Farm in Mecklenburg-Vorpommern.

Scenario	Soil organic carbon without cover crops (t CO_{2eq}/ha)	Soil organic carbon with cover crops (t CO_{2eq}/ha)	Difference (t CO_{2eq}/ha)	Annual GHG mitigation potential* (kg CO_{2eq}/ha)
Standard	216	240	24	297
Best case	216	264	48	598
Worst case	216	216	0	0

Note: to transform CO_{2eq} to C, the CO_{2eq} value must be divided by 3.67.
Source: own estimation based on IPCC (2019a).

Total costs in transition stage from implementing the mitigation strategy

The costs in the transition stage from using cover crops after winter wheat in the Typical Farm are shown in Table 6.8. There is no variation in the costs across categories, thus, only the Standard case is shown. Seed costs increase by 18 USD/ha due to the additional purchase of the oat seeds. The expenditure on fertilizer is reduced by 11 USD/ha as a result of the prevented leaching from cover crops. This reduction in nitrogen losses lowers the urea rate in corn silage (16 kg N/ha).

Diesel and labor are increased by 6 and 3 USD/ha, respectively. This is the consequence of requiring an additional pass with the disk harrow to cut and incorporate the residues from the cover crop. Machinery costs - depreciation and repairs - are increased due to the purchase of the seed drill as well as the additional pass with the disk harrow. The increase in this category is 7 USD/ha.

The categories of pesticides, liming, contractor and foregone revenue are not affected by the mitigation strategy; thus, these have no changes.

Using cover crops increases the cost in the Typical Farm by 24 USD/ha in the standard, best and worst cases and this value is used to calculate the mitigation cost of the strategy.

Table 6.8 Wheat costs with cover crops in Typical Farm in Mecklenburg-Vorpommern in transition stage (USD/ha)

	Baseline scenario	Standard case	
		Strategy	**Change**
Seed	60	78	18
Fertilizer	207	196	-11
Diesel	87	94	6
Labor	80	83	3
Machinery	167	174	7
Remaining	227	227	0
Total	**828**	**852**	**24**

Source: own estimation.

Emissions per hectare in transition stage from implementing the mitigation strategy

Table 6.9 presents the changes in GHG emissions in the transition stage from using oats as cover crops. The emissions in the standard, best and worst cases are the same in all categories except for land use.

The emissions from the category direct increase by 38 kg CO_{2eq}/ha as a result of the additional biomass from the cover crop. This biomass is decomposed in the soil and used as a substrate by soil microbes, which release N_2O as part of their metabolism. A minimal reduction is estimated for the emissions from indirect, which includes leaching and volatilization of nitrogen. This decrease of 1 kg CO_{2eq}/ha also can be attributed to the oat's biomass, which increases the nitrogen the systems lose as N_2O (direct emissions), implying that less nitrogen is available to be volatilized and leached (see chapter 3.5.1.3). The emissions from diesel are increased by 20 kg CO_{2eq}/ha because of the additional tillage pass.

The category "land use" has negative emissions, depicting the carbon sequestration potential achieved from the additional residues from the cover crop. This potential results in a reduction of 297 kg CO_{2eq}/ha in the standard case and 597 kg CO_{2eq}/ha in the best case. In the worst case, IPCC coefficients indicate that no carbon sequestration occurs. Nonetheless, the likelihood of that scenario has been already discussed and can be considered low.

The category "prevented leaching" represents the decrease in GHG emissions stemming from the reduced urea input assumed in corn silage, which is a result of seeding cover crops. The categories included in this estimation represent the sum of direct, indirect, urea and manufacture emissions driven by the usage of this fertilizer. The total reduction achieved is 212 kg CO_{2eq}/ha in the three cases. It must be noted that this reduction is enough to compensate for the increased emissions in the category "direct," which are driven by the biomass of the cover crop. Thus, even in the worst case of land use (no carbon sequestration occurs), the strategy still results in a net emission reduction.

The emissions from urea and manufacture are included in the category "remaining," as these are not affected by the mitigation strategy.

The total reduction of GHG emissions from using cover crop is 451 kg CO_{2eq}/ha in the standard case, 751 kg CO_{2eq}/ha in the best case and 154 kg CO_{2eq}/ha in the worst case. These values are used for the calculation of the mitigation cost.

Table 6.9 GHG emissions in winter wheat with cover crops for Typical Farm in Mecklenburg-Vorpommern in transition stage (USD/ha)

	Baseline scenario	Standard case		Best case		Worst case	
		Strategy	Change	Strategy	Change	Strategy	Change
Direct	1,858	1,896	38	1,896	38	1,896	38
Indirect	162	161	-1	161	-1	161	-1
Diesel	282	302	20	302	20	302	20
Land use	0	-297	-297	-597	-597	0	0
Prevented leaching	0	-212	-212	-212	-212	-212	-212
Remaining	706	706	0	706	0	706	0
Total	**3,007**	**2,556**	**-451**	**2,256**	**-751**	**2,853**	**-154**

Source: own estimation.

Mitigation costs in transition stage

The costs to reduce GHG emissions in the transition stage (20 years) utilizing cover crops on the Typical Farm are 52 USD/t CO_{2eq} in the standard case, 31 USD/t CO_{2eq} in the best case and 153 USD/t CO_{2eq} in the worst case. The considerations of this mitigation strategy are presented at the end of this assessment.

Total costs in stable stage from implementing the mitigation strategy

Since the carbon sequestered in the soil can be released as emissions if the input of biomass is reduced, the Typical Farm must continue to seed the cover crop after the transition stage. No changes in the costs are assumed between the stages as the same practices are used. Hence, the same costs from the transition stage are assumed for the stable stage, which are presented in Table 6.8. Using cover crops in the Typical Farm in the stable stage implies an increase in costs of 24 USD/ha in the standard, best and worst cases.

Emissions per hectare in stable stage from implementing the mitigation strategy

The emissions in the stable stage in all categories are the same as in the transition stage (see Table 6.9), except for land use. The sequestration of carbon depicted in this category is assumed to be completely realized in the stable stage. Consequently, this value becomes zero, implying that no further net change takes place. Moreover, since this was the only emission category with different values in the three cases, the change in emissions in the standard, best and worst cases in the stable stage become equal. The estimated reduction in these cases is a net reduction of 154 kg CO_{2eq}/ha. Consequently, using cover crops on the Typical Farm still amounts to a reduction in emissions in the stable stage, although at a significantly lower magnitude than during the transition stage.

Mitigation costs in stable stage and considerations on the adoption of the strategy

The use of oats as cover crops in the stable stage implies a GHG mitigation cost of 153 USD/t CO_{2eq} in the three cases, as the assumptions among them do not change. The increase in costs compared with the transition stage is the result of the carbon sequestration potential being fulfilled.

The considerations on monitoring this type of strategy have already been covered in chapter 4.5.4 for the case study of the Typical Farm in Iowa. A significant difference compared with that case is that the policing and reporting systems regarding the usage of cover crop already are used in the current version of the CAP.

Hence, it can be assumed that low or no additional costs would be perceived by the controlling entity, as the same systems could be used and only their maintenance must be financed, which would occur even in absence of the mitigation strategy.

6.6 Combination of strategies (DE-C)

The approach to assess the implications of combining the mitigation strategies for the Typical Farm has been already discussed for the case study in Iowa (Chapter 4.6). The evaluation for Mecklenburg-Vorpommern follows the same principles utilizing the baseline scenario (DE-0) for the comparison. Hence, the possible interactions between the individual strategies must be addressed to determine limitations in the production of winter wheat with the combined strategies.

Regarding the optimization of the nitrogen rate in winter wheat (DE-1), no alteration in the operations, soil or residue management in the Typical Farm is assumed. The only additional task is the utilization of a drone to sense the nitrogen content before the last pass to apply UAN, which is conducted by a contractor. The use of oats as cover crops (DE-2) does entail changes to the equipment and an additional operation, but these occur after the harvest of winter wheat. Similarly, the volume of crop residues increases because of the oats, yet this does not affect corn silage yields, as discussed in the previous chapter. Consequently, no interactions generating competition for machinery or labor, or limitations affecting soil and residue management, are generated by combining the strategies

The costs and emission reductions from the combination of strategies are evaluated in the following chapter utilizing the same considerations discussed in the Iowa case study. Only the possible interactions between individual elements of the strategies are evaluated. Assumptions from the individual strategies not presented in this assessment are presumed to be unchanged. The standard and best cases of each strategy are matched with the corresponding case of the other strategy to establish the cases in the combination. Nevertheless, the worst case in the combination is omitted since this case is calculated for only one strategy. Thus, the worst case in the combination would equal the worst case of the cover crop mitigation strategy (DE-2).

Costs and mitigation potential of optimization of nitrogen rate and cover crops combined

The combination of strategies reduces emissions by lowering the nitrogen input as well as by promoting carbon sequestration. Since this potential is limited in time, the costs and mitigation of this strategy are divided into a transition and a stable stage.

The optimization of nitrogen rate (DE-1) in winter wheat is the only one of the two strategies that reduces the input of nitrogen (UAN) in this crop. Thus, the fertilizer reductions assumed in that strategy are adopted for the combination without adjustments (see Table 6.4). Moreover, the trials from LfL (2001) indicate that the nitrogen available after winter wheat likely would not be affected by the decreased UAN rate, as explained in chapter 6.5.2. Therefore, the potential nitrogen that oats can absorb, thereby preventing leaching, can be assumed to be unaffected by the decrease in the UAN rate. Consequently, the same reduction in the urea rate in corn silage is assumed (16 kg N/ha), implying the same savings in emissions and costs.

Cover crops are the only strategy that promotes carbon sequestration. Hence, the sequestration potential of the individual strategy (see Table 6.7) is utilized for the combination. As with the single strategy, based on IPCC (2019a), the initial 20 years denote the interval with carbon sequestration; i.e., a net increase of soil organic carbon. In contrast, the stable stage portrays the soil in its steady state, meaning no net change in soil carbon.

Considering that no interactions take place and the changes of the individual strategies are added without adjustments, the combination in this Typical Farm can be interpreted as the weighted average of all the strategies assessed.

Total costs in transition stage from implementing the mitigation strategy

The costs in the transition stage for the Typical Farm from combining the optimized nitrogen rate and oats as cover crops are presented in Table 6.10. In all the cost categories, the best and standard cases have the same change. The only distinction is the expenditure on fertilizer.

Seed costs are increased by 18 USD/ha from the additional oat seeds used to establish the cover crop. Fertilizer costs have a net reduction of 14 USD/ha in the standard case and of 22 USD/ha in the best case. This change is the sum of the reduction of UAN derived from nitrogen optimization in wheat and the reduction in urea from the prevented leaching. However, the variation between the cases is attributed only to the differing decreases assumed for UAN, as prevented leaching is assumed identical in both cases.

Diesel and labor are increased in total by 10 USD/ha due to an additional pass with the disk harrow to incorporate the oat residues into the soil. The cost of depreciation and repairs, depicted by the category "machinery," increase by 7 USD/ha because of this additional pass as well as by the purchase of the seed drill mounted on the cultivator. The cost of the contractor increases by 28 USD/ha because a drone is used before the last application of fertilizer.

The category "remaining" includes the costs of liming and pesticides, which are not affected by the combination of strategies.

The resulting change in costs from using the combination of strategies is an increase of 49 USD/ha in the standard case and 41 USD/ha in the best case. These are used to calculate the mitigation cost in the transition stage.

Table 6.10 Wheat costs with combination of strategies for Typical Farm in Mecklenburg-Vorpommern in transition stage (USD/ha)

	Baseline scenario	Standard case		Best case	
		Strategy	Change	Strategy	Change
Seed	60	78	18	78	18
Fertilizer	207	193	-14	185	-22
Diesel	87	94	6	94	6
Labor	80	83	3	83	3
Machinery	167	174	7	174	7
Contractor	0	28	28	28	28
Remaining	227	227	0	227	0
Total	**828**	**878**	**49**	**869**	**41**
Seed	60	78	18	78	18
Fertilizer	207	193	-14	185	-22

Source: own estimation.

Emissions per hectare in transition stage from implementing the mitigation strategy

The GHG emissions from winter wheat during the initial 20 years of combining the mitigation strategies are presented in Table 6.11.

The emissions from direct are lowered by the reduced UAN rate but are increased by the additional biomass from the cover crop. In the standard case, the decrease in UAN is not enough to compensate for the emissions from the cover crops and direct emissions increase by a 12 kg CO_{2eq}/ha. On the contrary, in the best case, which assumes the highest UAN reduction, the net change in this category is a reduction of 71 kg CO_{2eq}/ha, indicating that the UAN emissions savings are higher than the release driven by the biomass. Indirect emissions are decreased by 19 kg CO_{2eq}/ha and 79 kg CO_{2eq}/ha in the standard and best cases. The difference can be attributed mainly to the decrease in the UAN rate.

The negative emissions from prevented leaching depict the decrease provided by the reduced urea rate in corn silage. This equals a reduction of 212 kg CO_{2eq}/ha in both cases, which entails the prevented direct, indirect and urea emissions triggered by this fertilizer application as well as the carbon footprint from its manufacture.

The CO_2 emissions released from the chemical structure of UAN are lowered due to the reduced rate. The net change is a reduction of 3 kg CO_{2eq}/ha in the standard case and 12 kg CO_{2eq}/ha in the best case.

The emissions from the manufacture of fertilizer used in wheat are lowered by 9 kg CO_{2eq}/ha in the standard case and by 40 kg CO_{2eq}/ha in the best case. The variation is explained by the difference reduction potentials in the UAN rate resulting from the optimized nitrogen rate. The emissions from diesel increase by 20 kg CO_{2eq}/ha as a result of the additional tillage pass required for the incorporation of the crop residues. Land use emissions, representing the carbon sequestration potential of the soil, decrease by 297 kg CO_{2eq}/ha in the standard case and of 597 kg CO_{2eq}/ha in the best case.

The GHG mitigation in the transition stage of the combination of strategies on the Typical Farm is a net reduction of 507 kg CO_{2eq}/ha in the standard case and of 990 kg CO_{2eq}/ha in the best case. These values are used to calculate the mitigation cost in this stage.

Table 6.11 GHG emissions in winter wheat with combination of strategies for Typical Farm in Mecklenburg-Vorpommern in transition stage (kg CO_{2eq}/ha)

	Baseline scenario	Standard case		Best case	
		Strategy	Change	Strategy	Change
Direct	1,858	1,870	12	1,787	-71
Indirect	162	142	-19	82	-79
Prevented leaching	0	-212	-212	-212	-212
Urea	235	232	-3	224	-12
Manufacture	471	461	-9	430	-40
Diesel	282	302	20	302	20
Land use	0	-297	-297	-597	-597
Total	**3,007**	**2,499**	**-507**	**2,017**	**-990**

Source: own estimation.

Mitigation costs in transition stage

The 20-year transition stage of the combination of strategies offers a mitigation cost of 97 USD/t CO_{2eq} in the standard case and 41 USD/t CO_{2eq} in the best case.

Total costs in stable stage from implementing the mitigation strategy

The same practice changes in winter wheat explained for the transition stage are assumed for the stable due to the reversibility of the carbon captured in the soil. Therefore, the same cost differences are calculated for the stable stage (see Table 6.10).

The total cost change in the standard case is a net increase of 49 USD/ha in the standard case and of 41 USD/ha in the best case. These are used to calculate the mitigation cost in the stable stage.

Emissions per hectare in stable stage from implementing the mitigation strategy

The stable stage signifies that the soil organic carbon has reached a new balance and there is no net change in its content. As a result, the carbon sequestration potential becomes zero depicted in land use, which is the only difference in the emissions compared with the transition stage. All the remaining categories have the same values as shown in Table 6.11 as well as the same explanations. The GHG emissions reductions in the stable stage are 210 kg CO_{2eq}/ha in the standard case and 393 kg CO_{2eq}/ha in the best case.

Mitigation costs in stable stage and considerations on the adoption of the strategy

In the stable stage, the mitigation costs with the combination of strategies are 235 USD/t CO_{2eq} in the standard case and 105 USD/t CO_{2eq} in the best case scenario.

Monitoring and policing the combination of strategies on this Typical Farm could be accomplished using the existing reporting system for the BMEL (2021a) and CAP payments, as discussed in the assessment of the individual strategies. Arguably, the transaction costs in the combination would be equal to the sum of the transaction costs of the individual strategies, as no changes would be expected.

6.7 Policy advice: ranking of GHG mitigation strategies

Table 6.12 presents the mitigation cost of the strategies evaluated for the Typical Farm in Mecklenburg-Vorpommern. To facilitate the comparison between strategies, a weighted average of 100 years between the transition and stable stages is calculated, as previously explained in chapter 4.7. The worst case is omitted from this comparison because it applies to only one strategy. The ranking is based on the mitigation cost calculated for each strategy using the time perspectives presented in chapter 3.8. It is recommended that policymakers utilize the ranking of strategies in the short-term perspective because of the urgency to achieve the goals in the Paris Agreement by 2050 (see chapter 2.6).

Table 6.12 Comparison of costs in the standard and best cases on average for Typical Farm in Mecklenburg-Vorpommern (USD/t CO_{2eq})

Strategy	Code	Standard case	Best case
Optimization of nitrogen rate	DE-1	460	73
Cover crops	DE-2		
Transition stage (short-term)		52	31
Stable stage		153	153
100-year average (long-term)		110	86
Combination	DE-C		
Transition stage (short-term)		97	41
Stable stage		235	105
100-year average (long-term)		183	80

Source: own estimation.

In the standard cases in the short term (transition stage), cover cropping offers the lowest mitigation cost of the strategies evaluated (52 USD/t CO_{2eq}), followed by the combination of strategies (97 USD/t CO_{2eq}) and, last, the optimization of the nitrogen rate (460 USD/t CO_{2eq}). The same ranking is valid in the long-term perspective (100-year average), with the corresponding costs being 110 USD/t CO_{2eq}, 183 USD/t CO_{2eq}, and 460 USD/t CO_{2eq}, respectively.

Moreover, the same holds true in the short term assuming the best case. Reducing a ton of CO_2, in this case, costs 31 USD/t CO_{2eq} with cover crops, 41 USD/t CO_{2eq} with the combination and 73 USD/t CO_{2eq} with the optimization of the nitrogen rate. However, this ranking is reversed when the long-term best case is assumed. In this case, the mitigation cost is the lowest with the optimized nitrogen rate at 73 USD/t CO_{2eq}. The cost with the combination is 80 USD/t CO_{2eq} and, finally, the costliest strategy is cover cropping with 86 USD/t CO_{2eq}.

Taking into account the need to quickly abate emissions to reduce the effects of climate change, utilizing cover crops can be considered as the advisable strategy from a cost-efficiency viewpoint. This strategy is the preferred alternative in the short term in the standard and best cases.

6.8 Summary and main findings

The case study of the Typical Farm in Mecklenburg-Vorpommern comprises two mitigation strategies focused on winter wheat production and their combination.

(1) In the baseline scenario, the Typical Farm has three passes to apply nitrogen in winter wheat. The NUE is increased by adjusting the fertilizer rate in the last application. However, a contractor is hired to measure nitrogen content before the application to adjust the rate if necessary and lower the risk of reducing the yield. The worst case is omitted as the NUE in the baseline already is higher than the optimum NUE assumed.

(2) According to the new CAP 2023, winter cover crops may be one of the mandatory alternatives to protect the soil from erosion. Oats can be used as cover crops after the harvest of winter wheat and before corn silage. This promotes carbon sequestration and prevents NO_3 leaching, which enables a reduction in the urea rate in corn. New equipment and further tillage operations are required, but this strategy offers the most economical mitigation cost.

(3) The combination of strategies is considered feasible without limitations or further adjustments. Thus, the individual strategies can be implemented without further considerations. As a result, the mitigation cost is the second lowest in the short- and long-term perspective.

It can be argued that cover crops are the best alternative in the short run, as they offer the lowest mitigation cost in the standard and best cases.

7 Comparison across Typical Farms

In this chapter, the mitigation costs and potentials calculated in the Typical Farms are compared. This comparison is divided according to the two time horizons considered in this thesis: short-term and long-term. As previously explained, despite the results from this thesis portraying only the farm level, the comparison can be used to generate additional advice for policymakers, as strategies from different Typical Farms can present similarities. Likewise, the comparison can be used to prioritize the need for further research to expand on the findings and analysis from this thesis. Thus, similar advice or solutions could be implemented in two or more Typical Farms with strategies with shared characteristics, indicating the possibility to cooperate. Afterwards, a comparison of the framework conditions in which the Typical Farms operate and how these could affect the implementation of the strategies is presented.

An overall finding from the case studies is that all the Typical Farms offer different numbers of feasible mitigation strategies. Four individual mitigation strategies are implementable in the Iowa Typical Farm, while three are possible in Paraná. Lastly, in Mecklenburg-Vorpommern, only two strategies are feasible. Additionally, in each Typical Farm, combining the strategies was feasible.

7.1 Mitigation potentials and costs

The first comparison in this chapter is the mitigation potential and corresponding mitigation costs from the three Typical Farms evaluated in the case studies. The individual strategies are compared first, including the description of differences and similarities across Typical Farms and mitigation strategies. The strategies are not ranked again, as this has already been done for each case study. The comparison is conducted for the combination of strategies as well.

Following the explanation in chapter 2.6 on the need for varying time horizons to provide advice to policymakers, the comparison is divided into short-term and long-term. The short-term perspective presents the strategies during the 20-year transition stage in which carbon is sequestered and the long-term perspective depicts the 100-year weighted average (see chapter 3.8). In each case, the first comparison is the mitigation potentials of the individual strategies, followed by the mitigation costs. The considerations valid in both perspectives are presented in the short term but are not repeated in the long term. It must be noted that none of the strategies in Paraná promote carbon sequestration; these are the same regardless of the time perspective considered.

A distinction is made for the Typical Farm in Iowa. In the case of the strategies that are implemented in corn and in soybeans, which are the reduction of tillage intensity (US-2) and cover crops (US-3), the average hectare was assumed as 50% corn and 50% soybean instead of only corn in the comparison, as already explained in chapter 4.7 for the definition of the ranking of strategies.

Carbon price as reference for competitiveness

Additionally, a reference carbon price is used to determine the competitiveness of the strategy's mitigation cost. This reference is the carbon certificate price (or carbon price), which depicts the economic cost an emitter must pay for the GHG emissions released. This price is derived from carbon markets, which operate on a cap-and-trade basis. In this system, an authority establishes a limited volume of emissions that can be released per year by a market or economic sector. These total emissions are divided into certificates representing a volume of emissions; e.g., one ton of CO_2. These certificates are allocated to the participants of the market that are obliged to have sufficient certificates to cover their total GHG emissions. The participants can trade the carbon certificates in the market. The carbon market encourages the adoption of GHG mitigation strategies, as long as these offer a lower mitigation cost than the carbon price.

This thesis takes place in the context of a research project in Germany, which participates in the European Emission Trading Scheme. This carbon market is among the first and largest in the world. However, as with most other carbon markets in the world, the agricultural sector is not mandated to participate in it (World Bank, 2022). Nonetheless, the carbon price from this market is used as a reference for this study. In this regard, the carbon price trend is assumed to be approaching 100 USD/t CO_{2eq} (World Bank, 2022). Moreover, this carbon price is also usually utilized in scientific literature, such as in the works by Smith et al. (2008), Pellerin et al. (2017) and IPCC (2014). Consequently, strategies with farm-level mitigation costs lower than this value are deemed competitive.

7.1.1 Short-term perspective

Mitigation potential of individual strategies

Figure 7.1 presents the mitigation potential calculated in the standard, best and worst cases of all the individual strategies assessed. The blue bar represents the total mitigation potential in the standard case as calculated in each strategy. This mitigation potential is divided into "Mitigation (no Land Use)" and "Land Use." The latter depicts the emission category with the same name and portrays the mitigation via carbon sequestration. This is presented separately to denote the share of this source of GHG reduction and relevance compared with strategies without it. The upper point in the chart depicts the best case and the lower point, the worst case.

Overall, the Typical Farm in Iowa offers the strategies with the highest mitigation potentials and Paraná the lowest. Moreover, there is a large variation in the mitigation potentials calculated in the case studies. This holds for the comparison within each Typical Farm as well as across them. To illustrate, the difference between the strategies with the highest and the lowest mitigation potential in the standard cases - i.e., cover crops in Iowa (US-3) and nitrogen efficiency in Mecklenburg-Vorpommern (DE-1) - is over 1,000 kg CO_{2eq}/ha.

Moreover, the potential from the same type of strategy can differ considerably. For instance, the optimization of the nitrogen rate in the Brazilian case study (BR-1) and in Iowa (US-1) are comparable, but these are five times larger than that for the German farm (DE-1). In principle, this difference indicates that the efficiency of nitrogen fertilization is comparatively the highest for the German Typical Farm in the baseline, as the emissions savings are the lowest. This high fertilizer efficiency in Germany can be attributed to the Fertilizer Directive (DüV), which determines the nitrogen rates that can be applied. Thus, taking into consideration only the mitigation potentials, this directive could be utilized as an example by policymakers to design similar policies for Brazil and the United States.

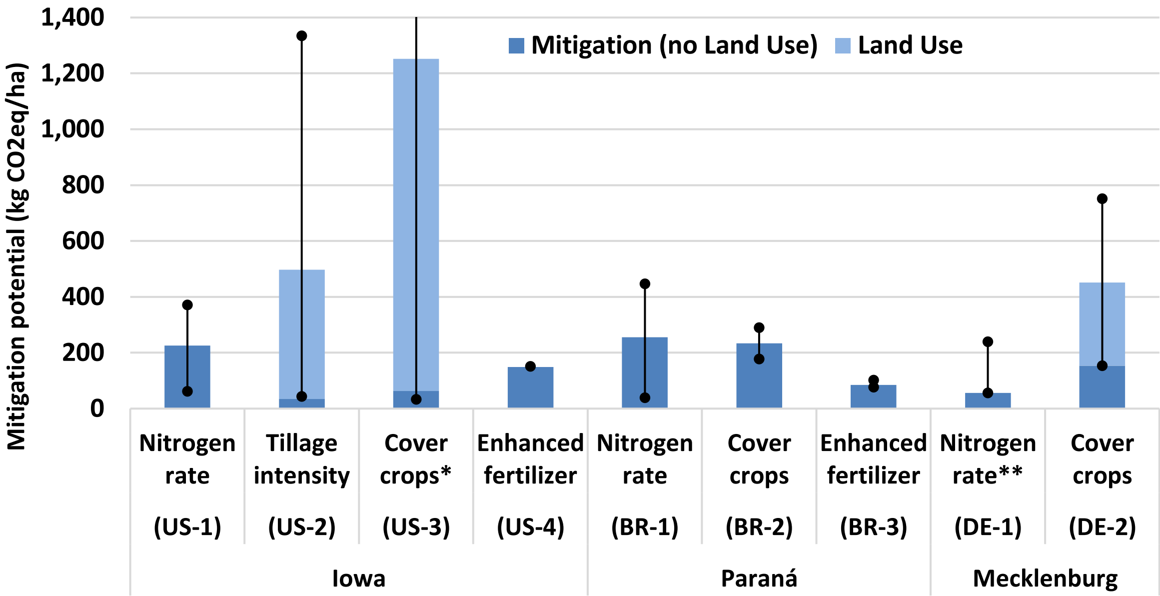
Reducing the tillage intensity offers the second-highest mitigation potential in Iowa (US-2), which is largely due to carbon sequestration. This is the only Typical Farm where this strategy can be implemented. The mitigation potential of cover crops can be significant, but it is not always the case. This strategy entails two mitigation approaches, the sequestration of carbon and the reduction of the nitrogen fertilizer. The reduction in the fertilizer rate occurs from nitrogen fixation and the prevention of leaching. The average hectare calculated for the Iowa Typical Farm (US-3) abates emissions with both approaches and has the highest mitigation potential of all the strategies assessed. Cover crops in Mecklenburg-Vorpommern (DE-2) reduce emissions by preventing nitrogen leaching and promoting carbon sequestration, yet this potential is only partly achieved due to limitations in the crop rotation, as the cover crop can only be grown in one of four seasons. The worst cases of cover crops in Germany and the USA have a noticeably low potential from the assumption that no carbon is sequestered, as already discussed in the respective chapters. Cover crops have been a common practice in Paraná (BR-2). Thus, the carbon sequestration potential is completed, and the strategy only lowers emissions by nitrogen fixation from legumes. The variation between the worst and best cases in Paraná is the lowest compared with the same practice in the other regions.

In general, strategies promoting carbon sequestration (US-2, US-3, and DE-2) have a markedly larger abatement potential compared with strategies that focus on decreasing nitrogen-driven emissions. This can be beneficial to attain a quick reduction of GHG emissions.

However, there is a significant variation between the best and worst cases in these three strategies compared with strategies only abating nitrogen emissions. The variation must be considered by policymakers because, while these three strategies may pose the largest mitigation potentials, it could be considered as unclear. While it is recognized that the methodology used to calculate carbon sequestration is not region-specific, which can partly explain these variations, there is an overall need to further research soil carbon dynamics to increase the accuracy of these calculations, which would also facilitate providing political advice (Amelung et al., 2020).

Enhanced-efficiency fertilizer (or inhibitors) in Iowa (US-4) has a mitigation potential of 150 kg CO_{2eq}/ha. This is almost twice as high as in Paraná (BR-3). This mitigation strategy in Brazil offers among the lowest abatement potentials from all the strategies studied. Still, in both cases, the strategy has a comparatively low variation among cases. Thus, despite the low mitigation potential, policymakers could perceive this strategy as a comparatively reliable approach to abate emissions.

Figure 7.1 Comparison of short-term mitigation potential for all Typical Farms



*Note: The mitigation potential in the best case is 2,481 kg CO_{2eq}/ha, but it is not shown to avoid distorting the chart.
**Note: The potential in the worst case is omitted from the analysis, as it is not possible to calculate it. This mitigation potential is displayed as equal to the standard case.
Source: own estimation.

Mitigation cost of individual strategies

Figure 7.2 presents the mitigation costs from all the mitigation strategies. The green bar presents the costs in the standard case; the bottom point of the line represents the costs in the best case; and the upper point, the worst case. The dashed red line represents the reference carbon price of 100 USD/t CO_{2eq} previously explained.

As with the mitigation potential, the mitigation costs vary substantially, ranging from -16 USD/t CO_{2eq} to 460 USD/t CO_{2eq} for the standard cases. The optimization of the nitrogen rate has negative mitigation costs (win-win scenario) in Iowa (US-1) and Paraná (BR-1). Together, these strategies represent the two most cost-efficient strategies in the standard case. However, in the worst case, they no longer are win-win scenarios. In contrast, in Mecklenburg-Vorpommern (DE-1), the strategy has the highest cost of all strategies evaluated, regardless of the case. As previously mentioned, this disparity could be attributed to the German Fertilizer Directive, as the mitigation potential in Mecklenburg-Vorpommern is comparatively low because of the policies already implemented. Based on these findings, it could be argued that policymakers in Paraná and Iowa should consider prioritizing programs aimed at increasing nitrogen efficiency. Despite the comparatively low mitigation potential compared with strategies promoting carbon sequestration, the negative costs indicate that the strategy not only is competitive, but its adoption could be facilitated because of the improvement in the economic performance. The mitigation costs in the German Typical Farm are markedly high; thus, in the economic sense, policies aiming to further reduce the fertilizer rate to lower emissions in this case would not be recommended. The transaction costs from monitoring this strategy could be considerable, as it is likely that farm visits are required to control the adoption of the strategy. Still, it can be expected that developments in satellite imagery makes the monitoring of the adoption of this strategy via remote sensing considerably cheaper.

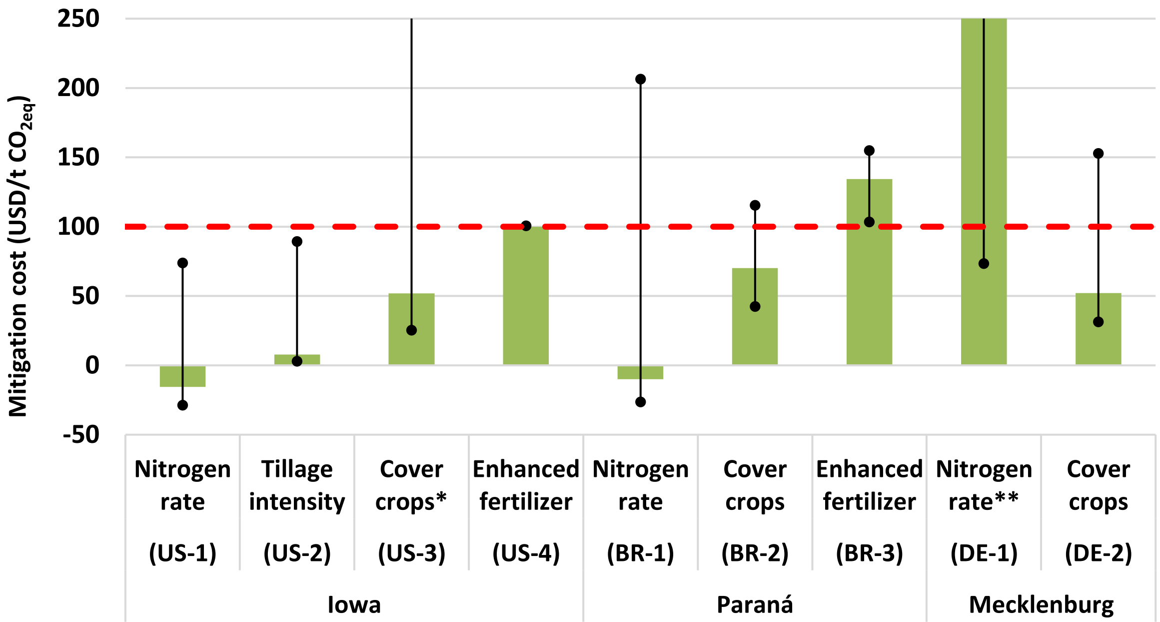
Reducing tillage intensity is the second cheapest scenario in the standard case in Iowa. Cover crops in the three farms offer a relatively similar mitigation cost, ranging from 50 to 70 USD/t CO_{2eq}. This occurs despite the considerable differences in the mitigation potential previously shown. However, the variation between the standard, best and worst cases is considerable. In the Iowa Typical Farm (US-3), the range is from 42 USD/t CO_{2eq} in the best case to 2,025 USD/t CO_{2eq} in the worst case. The high mitigation cost in the worst case is principally the result of the assumption of cover crops not sequestering carbon, as discussed in the respective assessment. Nonetheless, assuming the standard cases, the mitigation costs of these strategies could be considered competitive compared with the reference carbon price of 100 USD/t CO_{2eq}.

It must be noted that despite the comparably cheap mitigation potential of the strategies with carbon sequestration - i.e., tillage intensity in Iowa and cover crops in Iowa and Mecklenburg-Vorpommern - because of the uncertainties entailed in its estimation, portrayed by the variation between best and worst cases, their prioritization in policymaking should be critically evaluated (see previous section). For instance, further research to develop more accurate and region-specific models would help reduce the variation of the estimation, which, in turn, would support the definition of the ranking to provide advice to decisionmakers. Better estimations could reveal that the economics of strategies become considerably different, changing priorities of strategies. It could be expected that the transaction costs entailed in monitoring the adoption of the reduction of the tillage intensity and cover crops are comparatively low, as satellite imagery could be used, eliminating the need to visit the farm or conduct soil sampling. An exception in this regard is the monitoring of cover crops in Paraná. In this typical farm, a legume is added to the preexisting cover crop, implying that a mixture is used instead of a single plant species. Monitoring the adoption of the mixture would likely imply farm visits, as remote sensing is not always capable of differentiating between species. Thus, transaction costs could become significant. Nonetheless, improvements in this technology may make remote sensing a feasible alternative in the future, which would lower these costs.

Enhanced-efficiency fertilizers offer rather dissimilar mitigation costs in Iowa (US-4) and Paraná (BR-3). The costs are approximately 100 and 134 USD/t CO_{2eq}, respectively. The range between the best and worst cases in both Typical Farms is minimal. Nonetheless, the mitigation cost in Iowa is approximately the same as the reference carbon price. Thus, it could be considered competitive but there would not be any economic advantage from adopting it. Still, this strategy can be considered simple to adopt at the farm level and could be easily enforced; e.g., selling the fertilizers already mixed for the inhibitors, which would imply partly very low transaction costs. Moreover, in Iowa, the strategy shows low variation among the cases. Hence, while the farm-level mitigation cost may be the highest of all the strategies assessed in that Typical Farm, the ease of adoption and enforcement imply that this mitigation strategy potentially has other advantages. Arguably, the same could be concluded for Paraná, although the mitigation cost at farm-level is higher.

Overall, the Typical Farm in Iowa offers the most strategies with competitive GHG mitigation costs. Moreover, comparing the same mitigation strategy across the Typical Farms reveals that Iowa offers the lowest mitigation cost in all the mitigation alternatives assuming the standard case. The second cheapest mitigation potential can be achieved in the Paraná Typical Farm, although some strategies pose mitigation costs higher than the reference carbon price. In the case of Mecklenburg, only cover crops could be considered a competitive mitigation strategy. Contrary to the mitigation potentials, there is no clear trend between the source of mitigation (nitrogen-driven or carbon sequestration) and the costs calculated in each case study.

Figure 7.2 Comparison of short-term mitigation costs for Typical Farms

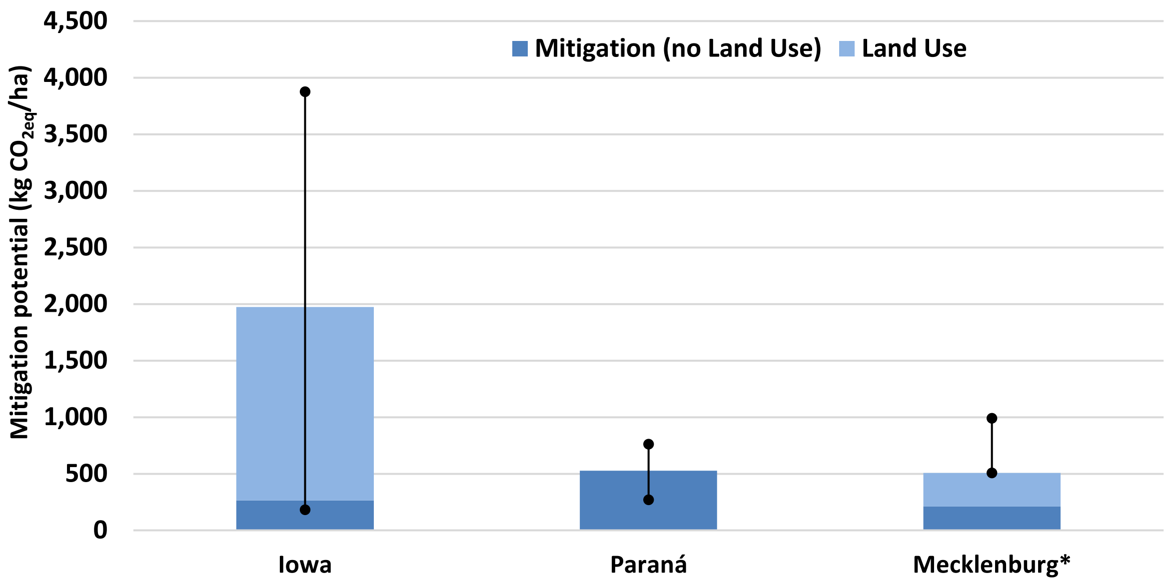


*Note: The mitigation cost in the worst case is 2,025 USD/t CO_{2eq}, but it is not shown to avoid distorting the chart.
**Note: The mitigation cost in the standard case is 460 USD/t CO_{2eq}, but it is not shown to avoid distorting the chart. The cost in the worst case is omitted from the analysis, but it is displayed as equal to the standard case.
Source: own estimation.

Mitigation potential in combination of strategies

Figure 7.3 presents the mitigation potential of the combination of strategies in each case study. The combined mitigation potentials in the Brazilian and German farms are similar in magnitude. In comparison, the Iowa Typical Farm has the largest mitigation potential of all the case studies, which is approximately four times larger than the other case studies. This can be attributed to two factors. The first is that the Iowa Typical Farm has the highest number of strategies that can be implemented simultaneously (four). The second is that two of the strategies promote carbon sequestration, which, as previously presented, offers a significantly larger abatement potential. However, the variation between the best and worst cases also is the highest in the Iowa Typical Farm. In this regard, the variation in the remaining two is deemed similar.

Figure 7.3 **Comparison of short-term mitigation potential from combination of strategies for Typical Farms**

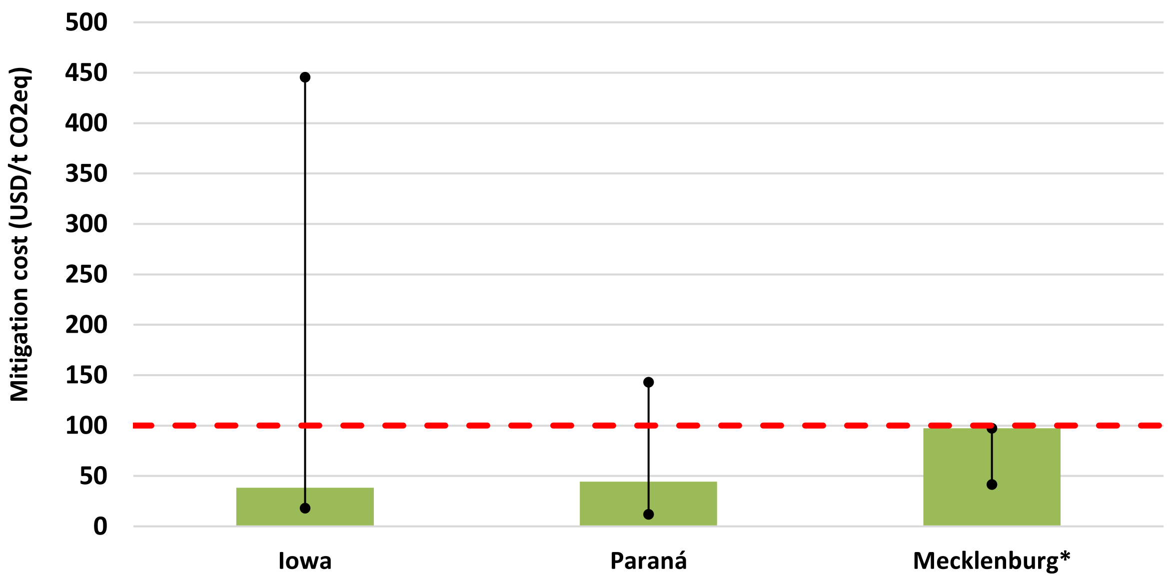


*Note: The potential in the worst case is omitted from the analysis, as it is not possible to calculate it. This mitigation potential is displayed as equal to the standard case.
Source: own estimation.

Mitigation costs in combination of strategies

The mitigation costs from the combination of strategies in the Typical Farms are presented in Figure 7.4. The mitigation cost in Paraná and Iowa are similar, although the variation between the best and worst cases in Iowa is considerably larger. The mitigation cost in the German case study is approximately twice as high in the standard case. Nonetheless, the mitigation costs of the combination of strategies in all the Typical Farms could be considered competitive for the standard case.

Figure 7.4 **Comparison of short-term mitigation costs from combination of strategies for Typical Farms**



*Note: The cost in the worst case is omitted from the analysis, but it is displayed as equal to the standard case.
Source: own estimation.

7.1.2 Long-term perspective

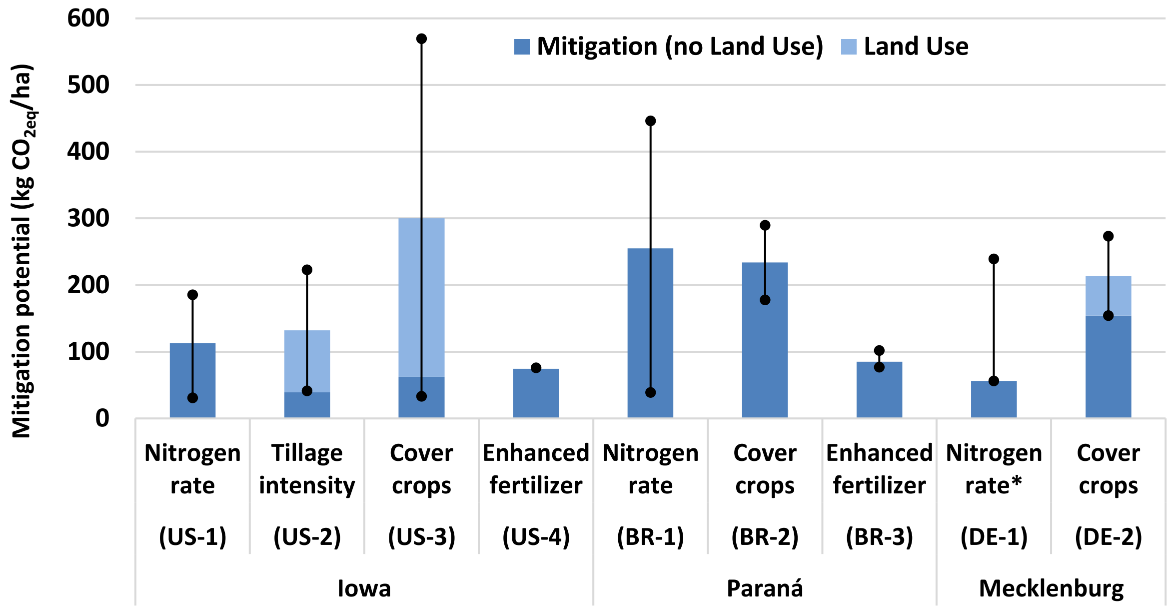
Mitigation potential of individual strategies

The long-term mitigation potentials for each Typical Farm are presented in Figure 7.5. The strategies that do not have carbon sequestration have the same mitigation potential and costs in the short- and long-term perspectives. These are not discussed again but are still presented to compare them with the strategies that entail carbon sequestration as these differ based on the time perspective. The strategies with carbon sequestration are the reduction of the tillage intensity and cover crops in Iowa (US-2 and US-3) and cover crops in Mecklenburg-Vorpommern (DE-2). Overall, in the long-term perspective, the variation of the mitigation potentials across the strategies is reduced compared with the short-term, but it is still significant.

The mitigation potential from transitioning to no-till (US-2) is distinctly lower in the long term than in the short term. This strategy's mitigation potential, in general, is comparable to some nitrogen-based strategies. Furthermore, it is less efficient than some of them;, e.g., the optimization of nitrogen efficiency in Brazil (BR-1). The mitigation via cover crops in the three farms is comparable in the long-term perspective. Thus, the contribution of the carbon sequestration to the overall mitigation of the strategy no longer is as substantial as in the short- erm. Cover crops in Paraná rely only on a reduced input of synthetic fertilizer resulting from nitrogen fixation to abate emissions. Still, this strategy in Paraná has a higher mitigation potential than in the German Typical Farm, which sequesters carbon. Nonetheless, the difference could be regarded as small. The variation between the best and worst cases is similar in the German and Brazilian Typical Farms, which could be regarded as low compared with the Iowa case study.

Overall, the strategies in the long-term perspective can be divided into two segments based on their mitigation potential. The first is composed of the strategies lowering emissions from 50 to 150 kg CO_{2eq}/ha, which includes strategies such as nitrogen optimization rate in Iowa (US-1) and Mecklenburg-Vorpommern (DE-1) and the enhanced-efficiency fertilizers (US-4 and BR-3). The second segment is the strategy abating emissions in the range of 200 to 300 kg CO_{2eq}/ha. Cover crops in all three case studies are in this range as well as the optimization of nitrogen rate in Paraná (BR-1). Thus, the strategies promoting carbon sequestration are not necessarily the only strategies offering a high mitigation potential compared with nitrogen-driven reductions, as was the case in the short-term. Moreover, the three Typical Farms have at least one mitigation strategy in each of these segments. Thus, unlike in the short-term where the Iowa farm offered a notably larger mitigation potential, all the Typical Farms have at least one strategy with a comparatively high mitigation potential. Hence, without accounting for costs, the evaluation of possible policies in this time perspective could not prioritize any specific type of strategy or source of mitigation potential equally across all Typical Farms.

Figure 7.5 Comparison of long-term mitigation potential for Typical Farms



*Note: The potential in the worst case is omitted from the analysis, as it is not possible to calculate it. This mitigation potential is displayed as equal to the standard case.
Source: own estimation.

Mitigation cost of individual strategies

The mitigation costs in the long-term perspective are presented in Figure 7.6. As with mitigation potential, all the results are presented, although only the strategies that promote carbon sequestration differ compared with the short term.

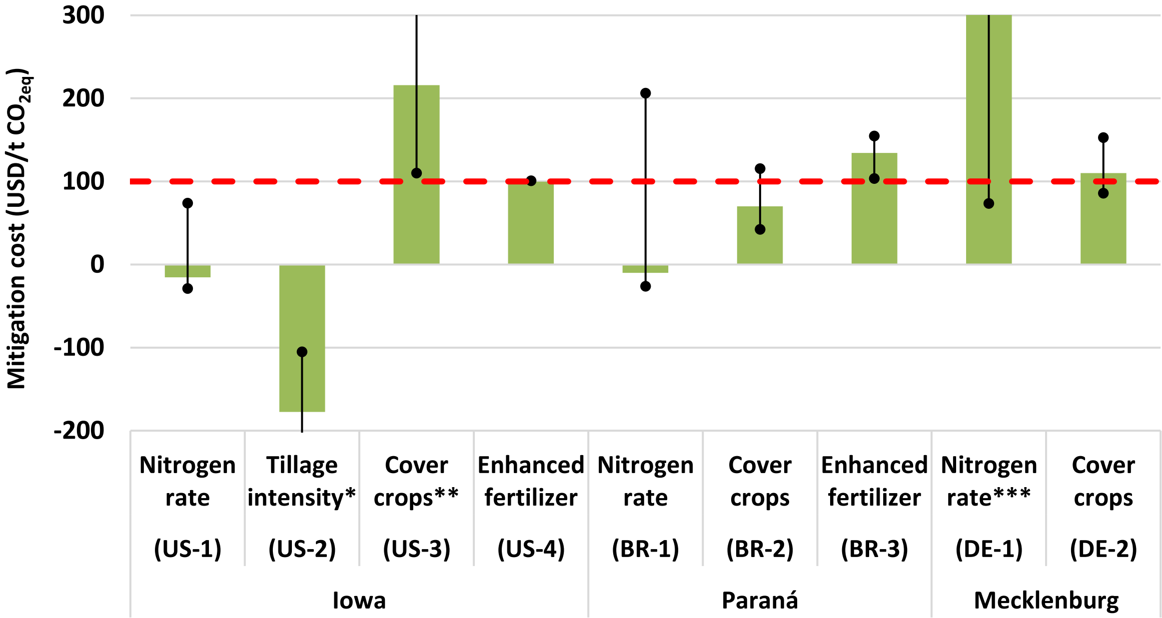
Considering all the strategies assessed, the variation across the mitigation costs in the standard cases is significant. This ranges from -177 USD/t CO_{2eq} to 460 USD/t CO_{2eq}.This also holds true for the variation of the worst and best cases of each strategy.

Contrary to the short-term perspective, in the long term, the reduction of tillage intensity (US-2) has a negative mitigation cost, but it also has the largest variation among all strategies. However, it must be noted that this strategy, as presented in the chart, must be interpreted differently. In the previous charts comparing mitigation costs, the bottom point depicted the best case and the upper one the worst case. However, due to how the mitigation costs are calculated (see chapter 3.7), the interpretation, in this case, is the opposite. This happens because both cases (best and worst) result in the same negative cost per hectare (numerator), but the mitigation potential in the worst case is smaller (denominator). Consequently, the result of the division in the worst case generates a greater mitigation cost than in the best case. This means that for this strategy, the bottom point, which, in this case, is not shown but has a value of -564 USD/t CO_{2eq}, presents the worst case, and the upper point (-177 USD/t CO_{2eq}), the best case. For policymakers in Iowa, this implies that policies aimed at promoting no-till should be prioritized over increased of the optimization of the nitrogen fertilizer rate (US-1). Still, given the unclarities on the calculation of the soil carbon sequestration and the transitory yield penalty explained in the strategy (see chapter 4.5.3), focusing on no-till as a policy would imply additional challenges beyond the farm-level mitigation cost.

Cover crops in Paraná (BR-2) in the standard case are the cheapest not win-win scenario of all the strategies. This is the only Typical Farm in which this strategy can be considered competitive in the long term, unlike in the short-term which was competitive for all three Typical Farms. Mitigation costs with cover crops in the Mecklenburg-Vorpommern Typical Farm and Iowa are markedly higher. As opposed to the short-term perspective, in the cases of Iowa and Mecklenburg-Vorpommern, this strategy no longer would have such a high priority to be promoted as policy based on the farm-mitigation cost. While, in the short-term, the uncertainties of the carbon sequestration potential could be perceived as balanced with the low cost in the standard case, this cost advantage is absent in the long term. Thus, strategies enforcing the adoption of enhanced fertilizers (inhibitors) can be considered as better policies than cover crops because of the low variation between the best and worst cases and, in Iowa, also lower costs.

Arguably, none of the strategies that differ in the long-term compared with short-term present a general trend regarding mitigation costs. In the long-term perspective, the strategies present considerable differences between the cheapest and costliest mitigation strategies. Similarly, no relationship between mitigation cost and source of mitigation (nitrogen-driven or carbon sequestration) can be derived for the long term. Nonetheless, the Iowa Typical Farm can still be deemed to offer the lowest mitigation costs, as it has two of the three cheapest mitigation strategies.

Figure 7.6 Comparison of long-term mitigation costs for Typical Farms



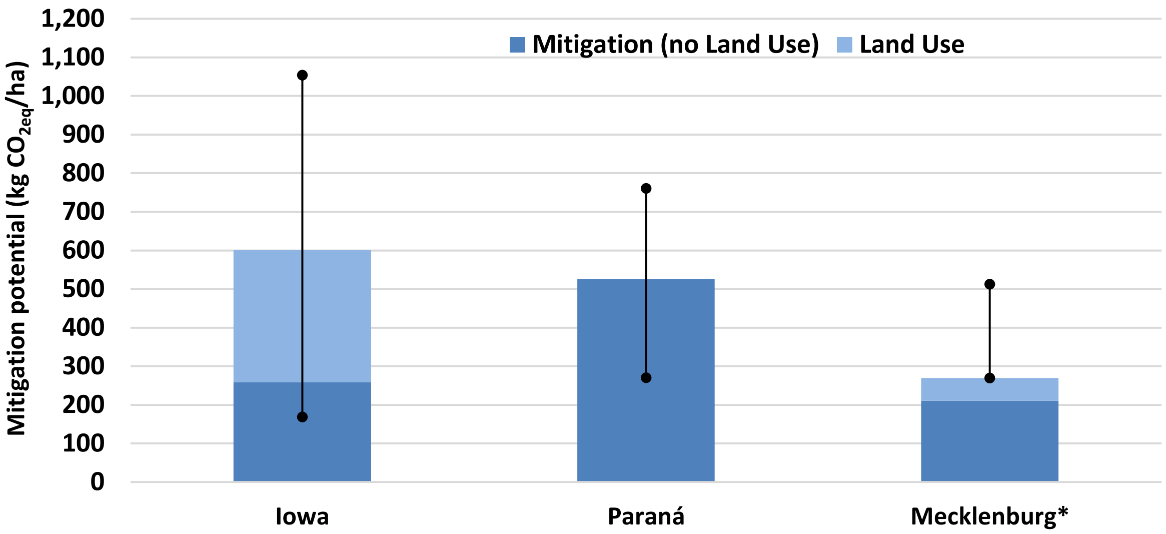
*Note: The mitigation in the worst case is -564 USD/t CO_{2eq}, but it is not shown to avoid distorting the chart.
**Note: The mitigation cost in the worst case is 2,025 USD/t CO_{2eq}, but it is not shown to avoid distorting the chart.
***Note: The mitigation cost in the standard case is 460 USD/t CO_{2eq}, but it is not shown to avoid distorting the chart. The cost in the worst case is omitted from the analysis, but it is displayed as equal to the standard case.
Source: own estimation.

Mitigation potential in combination of strategies

The mitigation potential in the combination of strategies in the long term is shown in Figure 7.7. The potential in the Typical Farm in Iowa and Paraná could be regarded as similar, especially if compared with the Mecklenburg-Vorpommern case. On average, this Typical Farm offers a potential only half as large as in Iowa or Paraná, if the standard cases are assumed. However, assuming the best case of each Typical Farm indicates that the Iowa Typical Farm has the largest potential, followed by Paraná and, lastly, Mecklenburg-Vorpommern. Thus, the results vary considerably based on the case assumed. Nonetheless, as in the short-term perspective, the variation between worst and best cases in Iowa is the largest.

As previously discussed, in the short-term perspective, the Iowa case has the highest combined mitigation potential, which resulted from having two strategies promoting carbon sequestration. Since carbon sequestration occurs in only 20 of the 100 years used in the weighted average for the long term, this potential is spread. Thus, the total potential becomes similar to the Brazilian case, which relies only on nitrogen-driven GHG mitigation.

Figure 7.7 Comparison of long-term mitigation potential from combination of strategies across Typical Farms

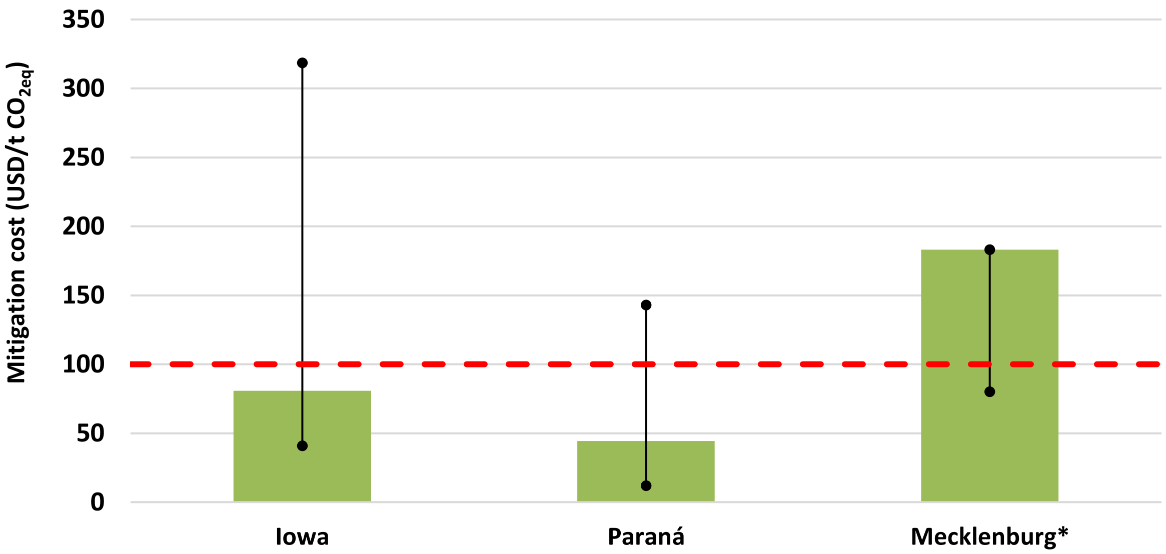


*Note: The potential in the worst case is omitted from the analysis, as it is not possible to calculate it. This mitigation potential is displayed as equal to the standard case.
Source: own estimation.

Mitigation costs in combination of strategies

The mitigation cost of the combination of strategies in the long-term perspective is presented in Figure 7.8. Assuming the standard case, the mitigation costs for the Typical Farm in Paraná are the lowest, followed by Iowa. The costs in Mecklenburg-Vorpommern are the highest and is the only Typical Farm where the combination of strategies is not competitive in the long term, unless the best case is considered.

Figure 7.8 Comparison of long-term mitigation costs from combination of strategies across Typical Farms



*Note: The cost in the worst case is omitted from the analysis, but it is displayed as equal to the standard case.
Source: own estimation.

7.2 Framework conditions for adoption and implementation of the strategies

Table 7.1 presents a summary of considerations on the context in which the mitigation strategies would be adopted in the Typical Farms. These are based on the comments by the participants from the focus group and the literature evaluated in the assessment of the individual strategies. It must be noted that this comparison is based on the framework conditions that decision-makers would need to consider for the implementation of the mitigation policies in the region. Therefore, aspects such as the uncertainties concerning soil carbon dynamics are not presented, as these elements are deemed intrinsic to the strategy and do not depend on the specific context in which the Typical Farm operates.

Table 7.1 Considerations on the adoption of the strategies and the regional institutional context

Typical Farm	Supporting institutions (knowledge and technical advisory)	Distribution of subsidies and economic support for grower	Enforcement, policing and monitoring of strategies
Iowa	Extension services (university and from state institutions); cooperatives	Systems for the distribution of subsidies already exist for cover crops but the system must be expanded if further mitigation policies are established	Currently only conducted for subsidized cover crops; expansion to monitor remaining strategies is necessary (medium transaction costs)
Paraná	Cooperatives	No systems in place (high transaction costs from their implementation)	No systems in place (high transaction costs from their implementation)
Mecklenburg-Vorpommern	Extension services (state institutions)	Systems for the distribution of subsidies (CAP) already exist for most elements in the strategies	Reporting system for growers already exists for the CAP and Fertilizer Directive, although adjustments may be necessary (low transaction costs)

Source: own estimation.

It could be argued that the transaction costs resulting from the implementation of the mitigation policies would be the lowest in Mecklenburg-Vorpommern. Due to the current regulations - i.e., Fertilizer Directive and CAP - reporting systems already are in place. Every year, growers must provide data on their application of fertilizer and acreage dedicated to specific crops, among other information. Thus, in a scenario in which the mitigation strategies would be implemented, these same systems could be expanded to include the necessary changes from the mitigation policy. This would be advantageous for the growers as well, as they already are familiar with the systems, so the additional knowledge required to manage and report the data would be reduced. Moreover, extension services, researchers and experts already are acquainted with the current systems. Additionally, based on the focus group discussion, the farm-level changes entailed in the adoption of the mitigation strategies are being researched in cooperation with growers. Consequently, growers would have access to region-specific data, likely without a considerable delay, as the expertise already is being developed before the adoption of the mitigation policy.

In comparison, policymakers in Iowa would need to consider that mitigation policies would require the establishment of new systems, implying comparatively higher transaction costs. There are programs available that provide economic support to growers utilizing cover crops, indicating that, at least for this strategy, this system could be expanded. However, this system is voluntary. Thus, in a scenario in which cover crops are mandatory, it is likely that considerable investment would be necessary to expand the reach to more growers. Moreover, while regulations regarding the usage of nitrogen fertilizer exist, growers do not have to report their fertilizer applications in detail. Similarly, they are not required to report their tillage management or use of inhibitors. Thus, for all strategies except cover crops, a system for reporting, policing and enforcing the policies would be necessary, entailing additional transaction costs. Nonetheless, the availability of experts from the USDA and the Iowa State University (extension services) to provide knowledge to growers was mentioned in the focus group discussions, especially considering that they currently conduct trials and make information available to growers. Additionally, the possibility to join a cooperative to have access to more consultancy services was considered relevant, although that would entail costs for the grower.

Lastly, it could be expected that the transaction costs entailed in mitigation policies in Paraná would be the highest of the three case studies. As indicated by the growers in the focus group, there are no systems providing economic support, nor requirements to report on their activities concerning the production of their crops. Therefore, the complete system for the application of the mitigation policy would need to be designed and developed, likely resulting in high mitigation costs. Moreover, they emphasized the lack of sufficient extension services from the state or universities, which they perceive as crucial to facilitate farm-level adoption of the mitigation strategies. The cooperative in which the Typical Farm participates conducts trials and distributes the results among the members, but these were regarded as not sufficient because of the limited scope, focusing mainly on yields of new varieties of crops. Hence, it could be expected that their contributions towards facilitating the adoption of mitigation strategies would be limited. Although extension services are not necessarily required for the implementation of mitigation policies, their services were considered key for their adoption at the farm level. Thus, establishing a network of extension services would further increase the costs derived from a mitigation policy, which should be accounted for by decision makers in the region.

8 Discussion

In this chapter, the application of the methodology and research concept as well as the limitations that it presents are discussed. The aspects discussed are valid for the entire thesis rather than for just an individual strategy on a Typical Farm, as strategy-specific discussions have been integrated into the corresponding assessment.

The Typical Farm concept and focus groups

Each case study is conducted by defining a Typical Farm to evaluate the initial situation as well as the implications of adopting the GHG mitigation strategies at the farm level. A significant share of the information and data for this assessment is provided by the growers and consultants participating in the focus group based on their own expertise. Quantitative data on the farm, such as acreage and yields, can be corroborated with statistical data, which is presented in the corresponding chapter. Yet qualitative information such as the type of nitrification inhibitor a grower would adopt or the utilization of specific equipment, which is essential to generating realistic results, is not verifiable because no database or previous research with this type of information is available for the regions studied.

Similarly, it is not guaranteed that any of the participants in the focus group have vast experience with the implementation of the mitigation strategies. While growers may make assumptions based on their suppositions and similar experiences, these may not completely depict all the necessary considerations for the adoption of the strategy. Furthermore, their assumptions may be based on biased or noncomparable information. Additionally, multiple growers could express opinions stemming from the same source of information, leading to a diminished discussion due to a reduced input of opposing views (O.Nyumba et al., 2018). In this regard, the findings and results from the focus groups were discussed with agricultural consultants and researchers in the respective regions to evaluate the need for corrections or further assessments. However, given that information gathered in the discussion is not covered in scientific literature, additional validation or correction of these limitations is not feasible.

A possibility to improve on this limitation would be to conduct multiple focus group discussions in each region with different participants. Contradicting results could then be used to promote further discussions among the groups. Alternatively, the number of participants in each session could be increased, although this could risk fragmenting the discussion into smaller clusters if the group becomes too large, making it more difficult to reach consensus (O.Nyumba et al., 2018). Both possibilities would help validate information for which no database exists. However, both alternatives imply increased costs and time requirements for the realization of the project, which were not available in the context of this thesis.

On average, focus group discussions lasted for four hours, including two pauses. In each session, the participation of the members of the focus group tended to decline over time, which mainly can be attributed to the overall length of the session. Arguably, this could signify that the first strategies were discussed more thoroughly than the latter ones. This was addressed after the first session by arranging the order in which the strategies are assessed based on the expected time required for the assessment in the focus group. Thus, strategies involving more complex changes to the baseline were discussed first. Additionally, more and longer pauses also helped with this issue. Nonetheless, dividing the discussion into two or more sessions could to more beneficial, which should be considered in future case studies.

An approach that helped mitigate these issues was the evaluation of the results of the focus groups with agricultural experts who did not participate in the sessions. In the majority of the cases, their expert opinion was supported by scientific literature, which was utilized in this thesis. Additionally, their concerns were integrated in subsequent discussions and interviews to further address them.

Nonetheless, the overall concept of the Typical Farm and its focus group approach led to a comprehensive assessment of the GHG mitigation strategies in which key agronomic aspects of crop production were evaluated. This allowed an adequate depiction of a complex system and evaluation of interventions to it beyond what would have been possible utilizing only aggregated statistical data. This applies to three case studies, meaning that the overall approach is suitable for different agroclimatic contexts.

Evaluation of effect of the strategies on yields

A key assumption in the approach is the yield attained on Typical Farms, as it helps understand the baseline situation and evaluate the possible implications of the strategies. This information is derived from statistical data and comments from the participants in the focus group. The yield assumption is necessary to determine total GHG emissions, estimate the possibility to lower inputs such as nitrogen fertilizers and to quantify emissions from ILUC and additional costs as foregone revenue. Hence, this assumption is key in the calculations of costs and emissions.

The initial assumption in the model is constant yields. This assumption is evaluated for each mitigation strategy using scientific literature and is modified if necessary. However, in most cases, the results found in literature present only findings from field trials or measurements. Furthermore, in most cases, these results depict only the changes from a short period after the modifications to the system occurred; e.g., information only on the season following the transition to no-till. Considering the soil's relevance to yields and the time-scale in which changes occur in it, validating the representativity of the inferences from this thesis in the long-term, in general, is not completely feasible. Nonetheless, this limitation could be considered common in this type of research and is not unique to this thesis.

It must be noted that the quantity of scientific literature used to generate the assumptions differed considerably among mitigation strategies and Typical Farms. In general, the availability of applicable information was scarcest in Paraná, regardless of the language used to search for information.

An alternative to improve on this aspect is the utilization of crop growth models. This can estimate the plant's development and yields based on soil, plant and climate parameters that can be modified to predict changes in the system (Huang et al., 2019). Thus, the variables used in the models could be adapted to depict the conditions of the Typical Farms and simulate yields based on long-term changes. However, the models require regional field trial data to be calibrated to increase their accuracy (Kasampalis et al., 2018). This data is not available in every region, which indicates that there is the need to conduct additional research before models can be adopted. Additionally, the utilization of these models, in general, is a complex process that would require additional time and budget beyond this thesis.

Comparability of standard, best and worst cases for risk analysis

As described in chapter 3.6.1, the approach used to define the standard, best and worst cases in each strategy entails the combination of variables obtained from diverse sources using different methodologies. This limits the comparability of three cases because it is not feasible in every case to assign probabilities or statistical distributions to the variables without increasing the uncertainty of the assessment.

A possible improvement on this aspect for future research is the inclusion of expert knowledge to complete the missing statistical description of the variables. For instance, for cases with missing information, experts could provide an approximation on the type of statistical distribution or range of values that a variable may have. This would enable the utilization of statistical models such as Monte-Carlo simulations to calculate the uncertainty and sensitivity of the emission and cost estimation. However, expert knowledge may introduce inconsistencies in the analysis and increase the overall uncertainty of the results (Basset-Mens et al., 2009). In any case, considering the diversity in the types of data and regions evaluated in this assessment, gathering the data was not feasible in the context of this thesis because of budgetary constraints.

Variables and emission factors in the calculation of emissions

As explained in chapter 3.5, several coefficients and emissions factors used in the calculation of GHG emissions, especially in the case of the formulas obtained from IPCC (2019a), are disaggregated only by major climate and soil categories. Yet, this level of disaggregation could be assumed to encompass several subcategories, for which no information is provided. Consequently, these coefficients and factors can be considered to be an approximation of the conditions of the Typical Farms rather than an accurate depiction. Thus, these may not necessarily portray the local conditions of the farms. Furthermore, for some mitigation strategies, it was necessary to use data from other areas as no information from the exact region could be identified. While these other regions are considered comparable, differences may exist that can alter the result calculated for the case studies and cannot be accounted for. Hence, there is an intrinsic uncertainty in all the Typical Farms.

This aspect could be improved by developing regional variables specific to the conditions of the Typical Farms. For instance, trials in farms to measure the variables in the region could be established for this purpose. Moreover, these also could include further levels of disaggregation not previously considered, such as more specific types of nitrogen fertilizer, to account for more variation and increase accuracy. Additionally, models could be developed to adjust the variables and emission factors to the specific agroclimatic conditions of the Typical Farms. The development of these models must be accompanied by field trials for calibration and validation of results. Developed countries are working on improving models to increase their emission reporting, but these are not yet usually implemented (Albanito et al., 2017). In this regard, the United States, Brazil and Germany still report their emissions using the standard values provided by IPCC.

Relevance of emissions from transport of fertilizer and other inputs

The model for the GHG calculation does not account for the transport of the fertilizer from the factory to the Typical Farms. The model also assumes that the fertilizers applied in the Typical Farm are produced in the country where the farm is located. However, this is not necessarily the case, as a significant share of fertilizer may be imported from other countries or continents (Walling and Vaneeckhaute, 2020). This transport, regardless of the type, entails GHG emissions from fuel combustion and energy usage. Consequently, the GHG emissions assigned to each unit of fertilizer would be higher than what is assumed in this thesis if emissions from transport are accounted. Hence, the emissions from crop production would have been higher in the baseline scenario. Furthermore, this also implies that by lowering the input of fertilizers in the Typical Farm, the mitigation potential achieved is actually higher than what the results from this thesis indicate. In this regard, it was not possible to determine where the fertilizers are produced or how they are transported; thus, this effect is not quantifiable.

Nonetheless, the share of the emissions entailed in the transport of fertilizer can be regarded as low compared with manufacturing and application. In case studies conducted in Germany, Hasler et al. (2015) estimated that the emissions from the transport of fertilizer accounted for 1% to 3% while Kathrin et al. (2017) calculated them to account for less than 1% in the case of nitrogen fertilizers. Similar findings are provided by Zhang et al. (2013) that indicate the transport of nitrogen-driven fertilizer used in China accounts for 0.7% of the total. Consequently, it can be inferred that overall, the emissions entailed in the transportation of fertilizers, not included in this thesis, are rather irrelevant compared with the release from the manufacture and application. Thus, their omission can be assumed to have a negligible effect in the total mitigation cost.

Exchange rate effects in comparison of mitigation costs

All the cost calculations for the Brazilian and German farms are conducted in the local currency and are transformed into USD in the last step. Therefore, unlike the Typical Farm in Iowa that is already estimated in USD, the resulting costs from these two Typical Farms are affected by the exchange rate used. This affects the comparison of the mitigation costs across regions, indicating that the competitiveness of the strategies across Typical Farms could vary depending on the value of the currency assumed.

In general, the Brazilian currency can be considered weak (Palley, 2021). Thus, when converting the results from the Paraná Typical Farm into USD, the mitigation costs might be indicated to be cheap, especially compared with the German farm. Moreover, the exchange rate of the Brazilian real has been comparatively less stable than the Euro since the year 2000 (World Bank, 2021). For instance, taking the average of the exchange from 2010-2012 as opposed to 2016-2018, as assumed in this thesis, would almost halve the exchange rate (1.8 reals per USD compared with 3.4), roughly doubling the mitigation costs. Consequently, the overall ranking of mitigation cost across the Typical Farm could be different in the future derived from variations in the exchange rate.

Negative mitigation costs (win-win scenarios)

The optimization of the nitrogen rate in Iowa (US-1) and in Paraná (BR-1) as well as the reduction of tillage intensity in Iowa in the long term (US-3) are indicated to have negative mitigation costs, implying that these are win–win scenarios. Thus, the grower not only would lower emissions but would also improve economic performance by implementing the mitigation strategy. This result could be interpreted as counterintuitive, as it could be argued that growers operate as profit maximizers before the mitigation strategies are implemented. Hence, in theory, they would have already adopted the strategies as it would increase their profit.

However, there are considerations that may partly explain this particular result. Moran et al. (2013) argues that, in the context of research, an oversimplification of the production system and costs, utilization of nonrepresentative data or the adoption of statistical averages can lead to cost calculations that are not realistic. Further aspects such as the transaction costs associated with acquiring know-how may have been omitted in the cost estimation as these are extremely complex to generate (MacLeod et al., 2015). Vermont and Cara (2010) argue that most economic estimations have implicit inefficiencies and omitted costs. Hence, it could be claimed that the negative costs of the strategies are a result of the intrinsic limitations of the model and that the strategy, in reality, would not generate the economic savings indicated because of unaccounted costs.

In addition, it is possible that growers do not behave as profit maximizers in all instances, as they are influenced by cultural and personal factors such as tradition and risk perception, perception of the policy environment and business constraints (Pike, 2008). Moreover, growers' risk affinity can imply that they knowingly oversupply a specific input, such as nitrogen, expecting that favorable weather conditions generate a higher yield. Hence, they perceive the oversupply as a risky investment from which they would benefit by attaining a higher revenue. Consequently, it also is possible that the negative costs estimated are a realistic depiction, but the growers are not pursuing the economic benefit from the mitigation strategy due to other considerations. Nonetheless, win–win scenarios are a promising alternative that must be evaluated further, along with the influence of farmers' behavior to generate meaningful political advice.

9 Conclusions and advice for policymakers

The assessment of the GHG mitigation strategies designed for cropping systems indicates that the Typical Farm in Iowa offers the highest number of individual mitigation strategies (four), followed by Paraná (three) and lastly Mecklenburg-Vorpommern (two). Additionally, for each farm, it is possible to combine the strategies for a join implementation, which can be perceived as an additional strategy. Moreover, this thesis revealed there is a considerable variation in the GHG mitigation potentials and costs. This variation is present within the strategies, shown by the differences in the standard, best and worst cases, as well as between the strategies in the farm. Furthermore, in some cases, the mitigation potentials, and costs for a specific type of strategy - e.g., optimization of the nitrogen fertilizer rate - can differ significantly depending on the farm on which it is implemented.

This thesis calculated the mitigation potentials and costs of the strategies at the farm level. These mitigation costs are a key component for the design of efficient mitigation policies. However, they are not the only aspect that must be considered by policymakers, as strategies may present tradeoffs, such as a reduction in yields, which can generate indirect land use change. Moreover, the mitigation strategies entail different transaction costs for policing and enforcing them, which also depend on the specific design of the policy. Hence, the results calculated are not sufficient to provide definite advice for policymakers. Still, the findings can be used to select which strategies should be prioritized to conduct further analysis of transaction costs and to focus the need for further research.

Additionally, in the context of mitigation strategies in cropping systems, policymakers should differentiate between strategies offering an abatement of emissions and strategies promoting carbon sequestration. The abatement of emissions - e.g., optimize nitrogen fertilization, thereby lowering the rate to reduce emissions - can be realized every year, permanently mitigating emissions compared with the baseline scenario. Consequently, the potentials and costs of these strategies are not affected by the duration of the policy. In contrast, the soil's capacity to sequester carbon is limited and the process is reversible. Hence, the strategy that led to the sequestration must be used permanently, despite it no longer capturing additional carbon after the potential is fulfilled. As a result, strategies with carbon sequestration vary depending on the time horizon considered. Consequently, a comparison and ranking of the strategies generates different results based on the time perspective. Nonetheless, because of the need to quickly lower global GHG emissions to ameliorate the effect of climate change, it is recommended that a short-term perspective is selected to define mitigation policies.

Considering a short-term perspective, strategies promoting carbon sequestration are indicated to offer the largest mitigation potentials. However, these strategies, in general, present the largest variation between the best and worst cases, indicating that the actual mitigation achieved may differ considerably from the standard value assumed, potentially risking a largely different outcome. This variation is the result of the uncertainty in calculating carbon sequestration, which is partly attributable to the methodology utilized in this thesis. However, there is an overall need for more research in this field to produce more accurate soil models. These improved models in the future could quantify the mitigation potentials to generate more accurate results, reducing the variation shown in this thesis. In comparison, strategies that abate emissions without promoting carbon sequestration have a comparatively low mitigation potential, but they have a relatively low variation between the best and the worst cases, which can be perceived as an advantage. Nonetheless, additional research to generate region-specific data, such as yield curves, is still needed. In the long-term perspective, there is no clear distinction between the type of strategy and the potential it offers. Thus, no overall trends between the type of strategy and its potential can be identified.

Similarly, there are important differences in the mitigation costs across the strategies. In general, optimizing the nitrogen fertilizer rate presents negative mitigation costs in Iowa and Paraná. This indicates that, by adopting the strategy, emissions would be abated, and costs lowered, because of the reduction in the fertilizer rate. Yet, these win–win scenarios, which seem counterintuitive in an economic sense, are highly debated. Arguably, they could be the result of the limitations of the calculations used in this thesis because some costs may have been unaccounted for, and the real change may be an increase in costs. This strategy in Mecklenburg-Vorpommern is among the most expensive of all the case studies. Germany already has implemented regulations to the application of nitrogen fertilizers that have led to a high nitrogen-use efficiency. Thus, policymakers could utilize the German system as an example to design policies for the USA and Brazil. Nonetheless, while the strategy may offer negative costs at the farm level, making this strategy into an interesting potential policy, it is likely that the transaction costs can become significant, as regular soil samples may be necessary to police the adoption of the strategies. Still, remote sensing technology capable of measuring the nitrogen in the soil is being developed, which would significantly lower these transaction costs.

Using cover crops in Iowa and in Mecklenburg-Vorpommern as well as reducing the tillage intensity to no-till in Iowa promote carbon sequestration. In the short-term, these strategies offer among the lowest mitigation costs for their respective farms. However, due to the uncertainty in the estimation of the mitigation potential, they also can present a markedly large variation between the best and worst cases, such as in the case of cover crops in Iowa. It is likely that transaction costs from monitoring this strategy are comparatively low. Satellite imagery can be used to control the adoption of both strategies, which would make wide-scale controlling cheap. These low transaction costs can be interpreted as an advantage over strategies requiring soil sampling. Nonetheless, more research is needed to determine the magnitude of the difference in the transaction costs to provide adequate policy advice.

Cover crops in Paraná offer mitigation costs comparable to the same strategy in the other two cases. However, it differs as it abates emissions only by reducing the input of fertilizer because a legume is included in the cover crop already used. For this reason, the strategy has a lower variation compared to the same strategies in the other farms. Consequently, policymakers do not need to take into consideration the particularities of carbon sequestration and can compare it directly with the other strategies feasible on this farm. It is likely that farm visits are required to control the adoption of the mixture of cover crops, as remote sensing is not yet capable of reliably distinguishing between different species, meaning that transaction costs could be significant.

Inhibitors are feasible in Iowa and Paraná and have the lowest variation between the best and worst cases. This strategy offers some of the highest mitigation costs in the short-term, which, coupled with the lowest mitigation potentials, makes it arguably the least preferable strategy in the respective farms. However, they present two advantages compared with the other strategies. On a technical level, they are easy to adopt on the farm, as no changes in the operations or machines are needed because the fertilizer can be purchased already mixed with the inhibitors. Additionally, transaction costs are likely to be significantly lower than any other strategies, because it could be made mandatory that all fertilizers are sold already mixed, avoiding the need to control the farms. Consequently, while its mitigation potential and costs at farm level are poorer than the rest of the strategies, the ease of adoption and low transaction costs makes it a rather interesting alternative for policymakers.

A comparison of mitigation costs over the long term presents significant differences compared with the short term. The optimization of the nitrogen rate remains unchanged, indicating that the same conclusions can be drawn in this time perspective for this strategy. However, the strategies promoting carbon sequestration are considerably more expensive, even more than strategies not promoting sequestration. For instance, in Iowa, the mitigation costs with inhibitors are approximately twice as high as with cover crops in the short term, but in the long term, the costs of cover crops almost quadruple and become higher than fertilizers inhibitors. No further elements concerning transaction costs are affected in this regard. Consequently, unlike in the short term, where more trends could be identified to support policymakers, it is not possible to generate such conclusions in the long term.

This thesis assessed the limitations and synergies arising from combining all the strategies feasible for each farm. This combination maximizes the mitigation potential attainable on each farm and helps understand the contribution of the farms toward the fight against climate change. In the short term, Iowa offers the highest combined mitigation potential, which can be attributed to this farm offering the highest number of feasible strategies and having two strategies with carbon sequestration. In the long-term, the differences become less significant, but Iowa still profiles as the farm with the highest potential. Similarly, Iowa offers the lowest mitigation costs in both perspectives, with Mecklenburg-Vorpommern offering the highest. It must be noted that the transaction costs from combining strategies would be lower than the sum of the transaction costs of the individual strategies, as these are likely to have shared fixed costs. For instance, the same reporting and policing approaches could be used for more than one strategy - e.g., satellites to measure no-till and cover crops - or sharing the same reporting platform.

10 Extended summary

Mitigating anthropogenic greenhouse gas (GHG) emissions is essential to reduce the effect of climate change. The agricultural sector is not only severely affected by it but also is responsible for a significant share of the total emissions. Hence, it must have an active role in the fight against climate change. Nonetheless, research on GHG mitigation strategies has focused mostly on either field trials or statistical models.

Field trials usually evaluate the abatement of GHG emissions by strategy compared with a reference scenario. Yet, in most cases they do not assess the implication of adopting the strategy on a farm, such as changes in the costs or organizational repercussions. Consequently, the applicability of these findings in a realistic scenario is not properly accounted for. Statistical models evaluate the effect of the mitigation strategies, mostly on an entire sector. These models rely on statistical databases presenting averages of yields, inputs, and further variables, which are used to depict entire regions. Thus, the models are limited by the availability of data and rely on simplifying agricultural production to make inferences.

Still, crop production can be understood as a complex process that is affected by regional characteristics, such as weather and availability of inputs. In turn, these regional aspects determine the feasibility of the strategies in a farm and the corresponding mitigation potentials and costs, which can vastly differ across regions. In this regard, the inclusion of regional growers and agricultural experts can be perceived as essential in the assessment of the strategies. They not only possess expertise and information not portrayed in statistical databases but can also adequately evaluate the technical and agronomical implications of the strategies under the regional conditions in which the farm operates. Consequently, local expertise must be taken into account in order to derive realistic and comprehensive farm-level mitigation potentials and costs. Considering the limited time and resources, understanding which strategies are the most efficient is crucial to design efficient mitigation policies.

Against this background, this thesis studied the farm level mitigation potentials and costs of GHG mitigation strategies designed for crop production and evaluated their differences and similarities. Three case studies were conducted in different regions for a selected crop, representing diverse production systems and conditions. Crops and countries were chosen based on their importance in global agricultural production and food security. The possibility to cooperate with local scientists to support this project also was taken into account in this selection process. Within each country, a region was selected using the same principles. Based on these criteria, the case studies conducted were corn in Iowa (USA), corn in Paraná (Brazil) and wheat in Mecklenburg-Vorpommern (Germany).

For each of these regions, a Typical Farm was established utilizing the methodology defined by *agri benchmark*. Typical Farms depict the prevailing arable production system from the population of farms generating the bulk of the agricultural output. Focus groups with local growers and agricultural experts were conducted in each case to construct the baseline scenario of the Typical Farm. These were validated using statistical data.

Additionally, the farm-level adaptations required to implement the GHG mitigation strategies were discussed in the focus groups. Each strategy was introduced and discussed in the session until a consensus on the overall changes required in the Typical Farm was met. The strategies were assumed to be implemented independently from each other. The repercussions in the entire crop rotation were taken into consideration in this thesis. The results from the focus group discussions were validated using scientific literature. The likely approaches to monitor and enforce the adoption of each strategy, i.e., transaction costs, were discussed considering the results of the Focus Group discussion.

Primarily, the GHG emissions driven by nitrogen and soil management were calculated utilizing the standard methodology from the Intergovernmental Panel on Climate Change (IPCC). The emissions from the manufacture of fertilizers as well as from diesel usage are included as these have been shown to represent a considerable share of the emissions from cropping systems. In general, the calculation of emissions relies on coefficients depicting the losses from the system. These losses depend on the regional soil and climate conditions. However, these data for the emissions' calculation are rarely available at a region-specific level for all the variables necessary. Consequently, this thesis utilizes the default coefficients provided by IPCC. These, in part, are specific as they are divided into major soil and climate categories but can be considered to still lack specificity. Nonetheless, this limitation is common in this type of research and is not particular to this project.

The calculation of nitrogen losses and emissions revealed that IPCC's methodology does not properly depict the situation in the Typical Farm, which partially can be attributed to the lacking specificity of the coefficients. Utilizing this methodology would indicate that the Typical Farm is undersupplying nitrogen, which does not seem feasible as it would negatively influence long-term profitability. Consequently, it was deemed necessary to adjust the methodology to better depict the situation of the Typical Farms concerning nitrogen usage. This adjustment modified only a share of the total nitrogen emissions, but still implies that the results are not directly comparable with other calculations.

Moreover, there was a significant variation in the coefficients used to calculate GHG emissions and the mitigation potentials of the strategies. In most cases, these variations were presented as statistical ranges accompanying the default values. To depict this variation, each mitigation strategy was divided into three cases: standard, best and worst. The standard case could be considered as the average result expected from the mitigation strategy. The best case was calculated utilizing the assumptions that resulted in the highest mitigation potential, while the worst case led to the lowest mitigation potential possible. The coefficients used for these calculations were obtained from various sources that applied different methodologies and approaches. Hence, the comparability of these coefficients is limited, and it is not possible to allocate probabilities to the different cases. Nonetheless, the three cases provide an indication of possible results that can be expected.

A preliminary list of eight GHG mitigation strategies for crop production usually discussed in research was elaborated. The strategies can be divided into two categories depending on the approach to mitigate emissions: reduction of nitrogen-based emissions and carbon sequestration in the soil. All the strategies were evaluated for each Typical Farm utilizing a set of criteria aimed at determining the feasibility to generate realistic results. The first criterion defined was that the strategy must not be a part of the baseline of the Typical Farm. Second, the participants of the focus group must be at least somewhat acquainted with the strategy and the implications that its adoption can have. Third, there must be sufficient scientific data from the region to validate the assumptions required for the calculations. Last, the net increase in global GHG emissions due to indirect land use changes resulting from a decrease in yields on the typical farm must also be evaluated and dismissed. In theory, the mitigation strategies could negatively affect crop yields. This would trigger indirect land use change, which occurs when yields are lowered but the demand for the product remains unchanged. This foregone yield would be perceived as an economic signal for growers elsewhere to increase their output, which would trigger the transformation of land for agricultural use, generating significant GHG emissions.

Additionally, for each Typical Farm, the possibility to combine all the mitigation strategies considered feasible was studied. This is conducted by evaluating possible limitations and interactions that may occur from the changes required in each individual mitigation strategy.

Only four of the eight mitigation strategies were considered feasible in at least one of the Typical Farms. The Typical Farm in Iowa offers the highest number of possible mitigation strategies (4), followed by Paraná (3) and finally, Mecklenburg-Vorpommern (2). The adoption of the strategies shows similarities across the Typical Farms, but there are crucial differences derived from the equipment and availability of inputs in the farms. The key agronomic and technical changes obtained from the respective focus group discussions are:

- **Optimization of the nitrogen rate: i**n Iowa and Paraná, this strategy implies conducting one more pass to spread fertilizer. This is realized with grower's equipment in Iowa, but with a contractor in Paraná. In both cases, the savings from the reduced nitrogen rate are enough to compensate for the added costs entailed in the additional pass (machinery, labor and diesel or contractor). This implies that the strategy also increases the profit and, thus, is a win–win scenario.

  In Mecklenburg-Vorpommern, no additional pass is feasible, but a contractor is hired to measure the nitrogen remaining in the soil, enabling the grower to adjust existing fertilizer applications. Nonetheless, this amounts to a net increase in costs in this Typical Farm.

- **Reduction of tillage intensity:** The strategy is feasible only in Iowa and, to maximize sequestration potential, it is adopted in the entire crop rotation (corn and soybeans). Lowering the tillage intensity to no-till reduces expenditures on labor, diesel and machinery, as less operations are needed. A part of the tillage equipment can be sold as it is not used in any other operation in the Typical Farm. However, this strategy generates a yield penalty that implies transitory foregone revenue and emissions from indirect land use change, although these are lower than the reduction achieved.

- **Cover crops:** In Iowa, the winter cover crops vetch and rye can be seeded before corn and soybeans, respectively. Thus, the full potential of carbon sequestration can be attained and the fertilizer rate in corn can be lowered, as vetch can fixate nitrogen. The cover crop is terminated utilizing herbicides, which is considered an additional operation.

  Cover crops already are used in Paraná, but a mixture that includes vetch lowers the need to apply synthetic fertilizer. Insecticides are applied to the cover crop to ensure proper development.

  In Mecklenburg-Vorpommern, oats can be used as cover crops, but only a share of the sequestration potential is realized because of limitations in the crop rotations. Nonetheless, oats also lower the need to use fertilizers in the following crop. It is important to note that, because of changes in the regulations concerning the subsidies from the European CAP in Germany, it is expected that cover crops will become widely adopted, meaning that this strategy is likely to become a part of the new baseline.

  This strategy increased the expenditure for seeds in all Typical Farms, as well as in labor, diesel and machinery in Paraná and Mecklenburg-Vorpommern and contractors in Iowa. The costs of pesticides also are increased in Iowa and Paraná.

- **Enhanced-efficiency fertilizers (inhibitors):** Nitrification inhibitors can be used as an additive to the fertilizer utilized in Iowa. In Paraná, the fertilizer can be purchased already coated with urease inhibitors. In both cases, the inhibitors enable a reduction in fertilizer rate. No further changes to the farm operations are required. Utilizing the inhibitors increases the expenditures in fertilizers, which are partly reduced by the lowered fertilizer rate.

- **Combination of strategies:** Combining all the individual strategies was feasible in each of the Typical Farms without significant changes to the operations. In Paraná and Iowa, there are interactions between the strategies affecting the nitrogen supply, resulting in the need to calculate new fertilization rates. Moreover, a synergy leading to an increased carbon sequestration potential is identified in the Iowa Typical Farm. The costs for fertilizers, seeds, diesel, machinery and labor, and contractors increase in the three Typical Farms.

This thesis identifies that there is a time dimension to the economics of GHG mitigation strategies. The methodology assumes that the potential of the soil to sequester carbon is finite and occurs over 20 years. Furthermore, this process is reversible, indicating that the strategies that led to the sequestration must continue to be used, even after the potential is realized. Thus, the economic cost resulting from the adoption of the strategy must be covered, despite no longer sequestering carbon. Consequently, in the cases where carbon sequestration occurs, the mitigation potentials and costs of the strategies are evaluated in short-term and long-term perspectives. Strategies without carbon sequestration have the same potentials and costs in both time frames. Nonetheless, this finding reveals that the interpretation of the result from this thesis and overall, in this type of research, depend on the time horizon used as reference.

In general, the mitigation strategies are revealed to have significant variations in mitigation potentials. In some cases, a type of strategy can have markedly different results across Typical Farms. Moreover, the variation between the best and worst cases for some strategies also is substantial. The detailed comparison of the mitigation potentials in the case studies provided several conclusions:

- The optimization of the nitrogen rate offers similar abatement potentials in the Typical Farms in Iowa and Paraná (approximately 240 kg CO_{2eq}/ha). However, in Mecklenburg-Vorpommern, the same strategy presents a fifth as much reduction. This difference is attributed to the already high efficiency of nitrogen fertilization on the German farm, which implies that the reduction in the fertilizer rate is lower. The variation between the best and worst cases is comparable for the three Typical Farms
- The reduction of the tillage intensity, which is attainable only in Iowa, lowers emissions by 497 kg CO_{2eq}/ha. This value, which is considered high, is almost completely the result of the carbon sequestration potential.
- Cover crops offer markedly different mitigation potentials. In Iowa and Mecklenburg-Vorpommern, the strategy promotes carbon sequestration as well as a reduction in nitrogen-driven emissions. However, the potential in Iowa is three times larger than in Mecklenburg-Vorpommern (1,251 and 452 kg CO_{2eq}/ha). The difference is attributed to limitations in the crop rotation. Moreover, the potential in Iowa is the highest of all the individual strategies. In comparison, cover crops in Paraná lower GHG emissions by only 233 kg CO_{2eq}/ha. However, the variation between the cases is the lowest in Paraná. Nonetheless, cover crops are indicated to have the highest potential for each Typical Farm.
- The mitigation potentials from enhanced-efficiency fertilizer (inhibitors) are comparatively low. The potential in Paraná is 85 kg CO_{2eq}/ha. This is approximately 50% less than in Iowa and is among the lowest abatement potentials overall. Still, this strategy offers the lowest variation between the best and worst cases.
- In the short term, strategies with carbon sequestration have a significantly higher abatement potential than strategies focused only on lowering nitrogen emissions. The three strategies with the highest mitigation potential promote it. However, while this potential is estimated to be significant in this thesis, there still is considerable uncertainty in the scientific literature regarding how high this potential is. As a result, these strategies have a higher variation among standard, best and worst cases.
- The comparatively higher mitigation potential of strategies with carbon sequestration holds true only in the short-term perspective. Assuming a long-term scenario, strategies that only abate nitrogen-driven emissions offer mitigation potentials comparable to strategies with only carbon sequestration.
- Overall, the highest mitigation potential with the combination of strategies is found in Iowa (1,973 kg CO_{2eq}/ha), driven mainly by carbon sequestration. This outcome could be expected, as this farm offers the highest number of feasible mitigation strategies as well as strategies with comparatively high abatement potential. The potential in Iowa is approximately four times higher than in the other two farms. However, this applies only in the short term. In the long term, Paraná and Iowa have similar potentials (600 and 526 kg CO_{2eq}/ha), with Mecklenburg-Vorpommern offering roughly only half as much.

As with the mitigation potentials, the mitigation costs depend on the time perspective as well as the cases assumed. The comparison of costs also shows similarities but also differences across the strategies. It must be noted that prioritizing mitigation strategies that offer low farm-level mitigation costs, as calculated in this thesis, is key in the design of efficient mitigation policies. However, possible transaction costs that can arise from the monitoring and enforcing the strategies must also be taken into account. Thus, the mitigation costs should be considered together with the transaction costs:

- Optimizing nitrogen rates in Iowa and Paraná result in negative mitigation costs of 16 USD/t CO_{2eq}.and 10 USD/t CO_{2eq}. The strategy offers some of the lowest mitigation costs overall in the short- and long-term perspective. On the contrary, the same strategy in Mecklenburg-Vorpommern, has among the highest costs across all the strategies (460 USD/t CO_{2eq}). In general, compared with the other strategies, the variation in costs across the cases could be considered moderate. Nonetheless, the monitoring of the strategy would entail considerable transaction costs. Farm visits to conduct soil sampling to measure the nitrogen are likely to be required until the remote sensing systems are sufficiently developed to become a feasible alternative.

- Reducing tillage intensity in Iowa entails low mitigation costs in the short term (8 USD/t CO_{2eq}). This strategy has the second-lowest mitigation costs. Moreover, the strategy is indicated to have negative costs in the long term (-177 USD/t CO_{2eq}), becoming the most cost-efficient strategy of all. This is the only case in which a strategy becomes more cost-efficient in the long term compared with short term. In the short term, the variation can be regarded as comparatively low, but it increases in the long term. The transaction costs from this strategy could be expected to be comparatively low, as satellite imagery could be used to monitor the adoption of the strategy. This aspect can be interpreted as advantageous, making this strategy more competitive for policymakers.

- Cover crops have comparable mitigation costs for the three Typical Farms assuming the short term. These cost 52 USD/t CO_{2eq} in Iowa and Mecklenburg-Vorpommern but are higher in Paraná, reaching 70 USD/t CO_{2eq}. However, the variation is markedly higher in Iowa than in the other two Typical Farms that is derived from the large variation in the abatement potential from carbon sequestration. Arguably, this strategy in the short-term offers the third-best mitigation cost-efficiency. As with the reduction of the tillage intensity, satellites could be used, implying low transaction costs in Germany and USA. The exception would be the case of Paraná, as farm visits would be required to monitor the adoption of the mixture of cover crops. Remote sensing is not always able to distinguish between different types of plants, but this may become feasible in the future.

- The mitigation costs with cover crops are considerably higher using the long term. In Mecklenburg-Vorpommern, these are 110 USD/t CO_{2eq} and in Iowa 216 USD/t CO_{2eq}. Thus, the mitigation costs with cover crops vary across Typical Farms in the long-term.

- Enhanced-efficiency fertilizers (inhibitors) abate emissions at a cost of 100 USD/t CO_{2eq} and 138 USD/t CO_{2eq} in Iowa and Paraná. These represent the least cost-efficient type of strategy compared with the individual alternatives studied but, in general, have lower variations between the cases than the other strategies. Arguably, the transaction costs from this strategy would be the lowest, as it could be made mandatory to sell the fertilizer already treated with the inhibitors. Thus, while the mitigation costs in the farm are in general the highest from all the strategies evaluated, the low transaction costs make this strategy a noteworthy alternative.

- The mitigation costs with the combination of strategies are approximately 40 USD/t CO_{2eq} in Iowa and in Paraná if the short-term is used as reference. Yet, the variation is higher in Iowa. In Mecklenburg-Vorpommern, the mitigation costs are approximately two times higher (78 USD/t CO_{2eq}).

In the long term, costs in Iowa and Mecklenburg-Vorpommern are 80 USD/t CO_{2eq} and 183 USD/t CO_{2eq}, roughly twice as high as in the short term.

- Overall, the Typical Farm in Iowa can be considered to offer the lowest mitigation costs overall, as it offers among the cheapest mitigation alternatives in the short and long term. Following the same principles, Paraná is the second-cheapest Typical Farm regarding mitigation costs and, last, Mecklenburg-Vorpommern.

A ranking of the strategies that should be prioritized based on their mitigation cost is provided in subchapter 7 of each case study. The rankings are divided by Typical Farm, time perspective and case.

It is important to note that this thesis calculated all the costs in the local currencies before converting the final result to USD. Hence, the exchange rate assumed in these calculations also influences the evaluation of the mitigation costs across the Typical Farms. Thus, while the costs in the national currencies may remain stable, the international comparison may still differ if the exchange rates are modified. This is particularly relevant in the case of the Paraná case study, as the Brazilian currency (Reais) historically has had considerable fluctuations compared against the USD. On the contrary, the Euro can be considered stable.

This thesis did not evaluate the interactions that may result from implementing just a selection of strategies at the same time, rather than all strategies as in the combination. However, considering that no significant restrictions were identified in the evaluation of the combination in any of the Typical Farms, no limitations are expected.

Overall, the utilization of focus groups to derive the farm-level changes necessary for the adoption of the GHG mitigation strategies can be considered suitable. It was effective to gather qualitative and quantitative information usually unavailable in scientific literature and databases. Moreover, it enabled the inclusion of expertise in the local context, which is necessary to derive representative and realistic results. Nonetheless, these results are still influenced by the knowhow and experience of the participants, which is part of the character of case studies.

The results from this thesis can be considered specific to the regions where the case studies are conducted. Even so, these can still be used as reference for comparable locations. Moreover, the insights gained from the focus group discussions on technical and agronomic implications at the farm level as well as the considerations needed to evaluate the potentials and costs can be used as guidelines for future research and policy advice.

11 References

Abdalla, M., Hastings, A., Helmy, M., Prescher, A., Osborne, B., Lanigan, G., Forristal, D., Killi, D., Maratha, P., Williams, M., Rueangritsarakul, K., Smith, P., Nolan, P., Jones, M. B. (2014) Assessing the combined use of reduced tillage and cover crops for mitigating greenhouse gas emissions from arable ecosystem. Geoderma 223-225:9-20. doi: 10.1016/j.geoderma.2014.01.030

Abdollahi, L., Munkholm, L. J. (2014) Tillage system and cover crop effects on soil quality: I. Chemical, mechanical, and biological properties. Soil Science Society of America Journal 78(1):262-270. doi: 10.2136/sssaj2013.07.0301

Abendroth, L. J., Woli, K. P., Myers, A. J., Elmore, R. W. (2017) Yield-based corn planting date recommendation windows for Iowa. Crop, Forage & Turfgrass Management 3(1):cftm2017.02.0015. doi: 10.2134/cftm2017.02.0015

Acuña, J. C. M., Villamil, M. B. (2014) Short-term effects of cover crops and compaction on soil properties and soybean production in Illinois. Agronomy Journal 106(3):860-870. doi: 10.2134/agronj13.0370

agri benchmark Cash Crop (2022) Main Result Database [Restricted Access]. Accessed: 25.01.2022

Akiyama, H., Yan, X., Yagi, K. (2010) Evaluation of effectiveness of enhanced-efficiency fertilizers as mitigation options for N2O and NO emissions from agricultural soils: meta-analysis. Global change biology 16(6):1837-1846. doi: 10.1111/j.1365-2486.2009.02031.x

Albanito, F., Lebender, U., Cornulier, T., Sapkota, T. B., Brentrup, F., Stirling, C., Hillier, J. (2017) Direct nitrous oxide emissions from tropical and sub-tropical agricultural systems - A review and modelling of emission factors. Scientific Reports 7:44235. doi: 10.1038/srep44235

Alexander, P., Rounsevell, M. D., Dislich, C., Dodson, J. R., Engström, K., Moran, D. (2015) Drivers for global agricultural land use change: The nexus of diet, population, yield and bioenergy. Global Environmental Change 35:138-147. doi: 10.1016/j.gloenvcha.2015.08.011

Al-Kaisi, M. M., Archontoulis, S. V., Kwaw-Mensah, D., Miguez, F. (2015) Tillage and Crop Rotation Effects on Corn Agronomic Response and Economic Return at Seven Iowa Locations. Agronomy Journal 107(4):1411-1424. doi: 10.2134/agronj14.0470

Al-Kaisi, M. M., Kwaw-Mensah, D. (2007) Effect of tillage and nitrogen rate on corn yield and nitrogen and phosphorus uptake in a corn-soybean rotation. Agronomy Journal 99(6):1548-1558. doi: 10.2134/agronj2007.0012

Al-Kaisi, M. M., Yin, X. (2004) Stepwise time response of corn yield and economic return to no tillage. Soil and Tillage Research 78(1):91-101. doi: 10.1016/j.still.2004.02.011

Al-Kaisi, M. M., Yin, X. (2005) Tillage and crop residue effects on soil carbon and carbon dioxide emission in corn-soybean rotations. Journal of Environmental Quality 34(2):437-445. doi: 10.2134/jeq2005.0437

Al-Kaisi, M. M., Yin, X., Licht, M. A. (2005) Soil carbon and nitrogen changes as influenced by tillage and cropping systems in some Iowa soils. Agriculture, Ecosystems & Environment 105(4):635-647. doi: 10.1016/j.agee.2004.08.002

Alves, B. J. R., Boddey, R. M., Urquiaga, S. (2003) The success of BNF in soybean in Brazil. Plant and Soil 252(1):1-9. doi: 10.1023/A:1024191913296

Alves, B. J. R., Zotarelli, L., Fernandes, F. M., Heckler, J. C., Macedo, R. A. T. de, Boddey, R. M., Jantalia, C. P., Urquiaga, S. (2006) Fixação biológica de nitrogênio e fertilizantes nitrogenados no balanço de nitrogênio em soja, milho e algodão. Pesquisa Agropecuária Brasileira 41(3):449-456. doi: 10.1590/S0100-204X2006000300011

Amado, T. J. C., Fernandez, S. B., Mielniczuk, J. (1998) Nitrogen availability as affected by ten years of cover crop and tillage systems in southern Brazil. Journal of Soil and Water Conservation 53(3):268-271

Amado, T. J. C., Santi, A., Acosta, J. A. A. (2003) Adubação nitrogenada na aveia preta. II - Influência na decomposição de resíduos, liberação de nitrogênio e rendimento de milho sob sistema plantio direto. Revista Brasileira de Ciência do Solo 27(6):1085-1096. doi: 10.1590/S0100-06832003000600013

Amelung, W., Bossio, D., Vries, W. de, Kögel-Knabner, I., Lehmann, J., Amundson, R., Bol, R., Collins, C., Lal, R., Leifeld, J., Minasny, B., Pan, G., Paustian, K., Rumpel, C., Sanderman, J., van Groenigen, J. W., Mooney, S., van Wesemael, B., Wander, M., Chabbi, A. (2020) Towards a global-scale soil climate mitigation strategy. Nature Communications 11(1):5427. doi: 10.1038/s41467-020-18887-7

Andersen, D. S., Pepple, L. M. (2017) A county-level assessment of manure nutrient availability relative to crop nutrient capacity in Iowa: spatial and temporal trends. Transactions of the ASABE 60(5):1669-1680. doi: 10.13031/trans.12417

Angers, D. A., Eriksen-Hamel, N. S. (2008) Full-inversion tillage and organic carbon distribution in soil profiles: a meta-analysis. Soil Science Society of America Journal 72(5):1370-1374. doi: 10.2136/sssaj2007.0342

Assunção, J., Gandour, C., Rocha, R. (2015) Deforestation slowdown in the Brazilian Amazon: prices or policies? Environment and Development Economics 20(6):697-722. doi: 10.1017/s1355770x15000078

Atris, A. M. (2020) Assessment of oil refinery performance: Application of data envelopment analysis-discriminant analysis. Resources Policy 65:101543. doi: 10.1016/j.resourpol.2019.101543

Audsley, E., Stacey, K. F., Parsons, D. J., Williams, A. G. (2009) Estimation of the greenhouse gas emissions from agricultural pesticide manufacture and use. Online: dspace.lib.cranfield.ac.uk/handle/1826/3913

Aulakh, M. S., Walters, D. T., Doran, J. W., Francis, D. D., Mosier, A. R. (1991) Crop residue type and placement effects on denitrification and mineralization. Soil Science Society of America Journal 55(4):1020-1025. doi: 10.2136/sssaj1991.03615995005500040022x

Bai, X., Huang, Y., Ren, W., Coyne, M., Jacinthe, P.-A., Tao, B., Hui, D., Yang, J., Matocha, C. (2019) Responses of soil carbon sequestration to climate-smart agriculture practices: A meta-analysis. Global change biology 25(8):2591-2606. doi: 10.1111/gcb.14658

Bakam, I., Balana, B. B., Matthews, R. (2012) Cost-effectiveness analysis of policy instruments for greenhouse gas emission mitigation in the agricultural sector. Journal of environmental management 112:33-44. doi: 10.1016/j.jenvman.2012.07.001

Baligar, V. C., Fageria, N. K., He, Z. L. (2001) Nutrient Use Efficiency in Plants. Communications in Soil Science and Plant Analysis 32(7-8):921-950. doi: 10.1081/CSS-100104098

Banerjee, M. R., Burton, D. L., Grant, C. A. (1999) Influence of urea fertilization and urease inhibitor on the size and activity of the soil microbial biomass under conventional and zero tillage at two sites. Canadian Journal of Soil Science 79(2):255-263. doi: 10.4141/S97-049

Basset-Mens, C., Kelliher, F. M., Ledgard, S., Cox, N. (2009) Uncertainty of global warming potential for milk production on a New Zealand farm and implications for decision making. The International Journal of Life Cycle Assessment 14(7):630-638. doi: 10.1007/s11367-009-0108-2

Benech-Arnold, R. L., Sánchez, R. A., Forcella, F., Kruk, B. C., Ghersa, C. M. (2000) Environmental control of dormancy in weed seed banks in soil. Field Crops Research 67(2):105-122. doi: 10.1016/S0378-4290(00)00087-3

Blanco-Canqui, H., Ferguson, R. B., Jin, V. L., Schmer, M. R., Wienhold, B. J., Tatarko, J. (2014) Can cover crop and manure maintain soil properties after stover removal from irrigated no-till corn? Soil Science Society of America Journal 78(4):1368-1377. doi: 10.2136/sssaj2013.12.0550

Blanco-Canqui, H., Lal, R. (2009) Crop residue removal impacts on soil productivity and environmental quality. Critical Reviews in Plant Sciences 28(3):139-163. doi: 10.1080/07352680902776507

Blombäck, K., Eckersten, H., Lewan, E., Aronsson, H. (2003) Simulations of soil carbon and nitrogen dynamics during seven years in a catch crop experiment. Agricultural Systems 76(1):95-114. doi: 10.1016/S0308-521X(02)00030-6

BMEL [Bundesministeriums für Ernährung und Landwirtschaft] (2021a) Verordnung über die Anwendung von Düngemitteln, Bodenhilfsstoffen, Kultursubstraten und Pflanzenhilfsmitteln nach den Düngeverodnung (DüV): Grundsätzen der guten fachlichen Praxis beim Düngen (Düngeverordnung—DüV) vom 26. Mai 2017 (BGBl. I S. 1305), die zuletzt durch Artikel 97 des Gesetzes vom 10. August 2021 (BGBl. I S. 3436) geändert worden ist. Online: www.gesetze-im-internet.de/d_v_2017/BJNR130510017.html

BMEL [Bundesministeriums für Ernährung und Landwirtschaft] (2021b) Verordnung zur Durchführung der im Rahmen der Gemeinsamen Agrarpolitik geltenden Konditionalität (GAP-KonditionalitätenVerordnung - GAPKondV) - Stand 26.11.21. Online: www.bundesrat.de/bv.html?id=0816-21

Bollero, G. A., Bullock, D. G. (1994) Cover cropping systems for the central Corn Belt. Journal of Production Agriculture 7(1):55-58. doi: 10.2134/jpa1994.0055

Bortolini, C. G., Silva, P. R. F., Argenta, G. (2000) Sistemas consorciados de aveia preta e ervilhaca comum como cobertura de solo e seus efeitos na cultura do milho em sucessão. Revista Brasileira de Ciência do Solo 24(4):897-903. doi: 10.1590/s0100-06832000000400021

Bouchet, A.-S., Laperche, A., Bissuel-Belaygue, C., Snowdon, R., Nesi, N., Stahl, A. (2016) Nitrogen use efficiency in rapeseed. A review. Agronomy for Sustainable Development 36(2). doi: 10.1007/s13593-016-0371-0

Bouwman, A. F. (1996) Direct emission of nitrous oxide from agricultural soils. Nutrient Cycling in Agroecosystems 46(1):53-70. doi: 10.1007/BF00210224

Bouwman, A. F., van Vuuren, D. P., Derwent, R. G., Posch, M. (2002) A global analysis of acidification and eutrophication of terrestrial ecosystems. Water, Air, and Soil Pollution 141(1/4):349-382. doi: 10.1023/A:1021398008726

Bowles, T. M., Atallah, S. S., Campbell, E. E., Gaudin, A. C. M., Wieder, W. R., Grandy, A. S. (2018) Addressing agricultural nitrogen losses in a changing climate. Nature Sustainability 1(8):399-408. doi: 10.1038/s41893-018-0106-0

Braccini, A. D. L. e., Stülp, M., Albrecht, L. P., Ávila, M. R., Scapim, C. A., Ricci, T. T. (2010) Agronomic traits and seed yield produced in the soybean-corn crop in succession cropping. Acta Scientiarum. Agronomy 32(4). doi: 10.4025/actasciagron.v32i4.8064

Brainard, D., Henshaw, B., Snapp, S. (2012) Hairy vetch varieties and bi-cultures influence cover crop services in strip-tilled sweet corn. Agronomy Journal 104(3):629-638. doi: 10.2134/agronj2011.0360

Bremner, J. M., Breitenbeck, G. A., Blackmer, A. M. (1981) Effect of nitrapyrin on emission of nitrous oxide from soil fertilized with anhydrous ammonia. Geophysical Research Letters 8(4):353-356. doi: 10.1029/GL008i004p00353

Brentrup, F., Joachim Lammel, Tiffanie Stephani, Bjarne Christensen (2018) Updated carbon footprint values for mineral fertilizer from different world regions. Online: www.researchgate.net/publication/329774170

Brentrup, F., Küsters, J., Lammel, J., Barraclough, P., Kuhlmann, H. (2004) Environmental impact assessment of agricultural production systems using the life cycle assessment (LCA) methodology II. The application to N fertilizer use in winter wheat production systems. European Journal of Agronomy 20(3):265-279. doi: 10.1016/S1161-0301(03)00039-X

Brentrup, F., Palliere, C. (2010) Nitrogen Use Efficiency as an Agro-Environmental Indicator. Online: www.oecd.org/greengrowth/sustainable-agriculture/44810433.pdf

Broch, A., Hoekman, S. K., Unnasch, S. (2013) A review of variability in indirect land use change assessment and modeling in biofuel policy. Environmental Science & Policy 29:147-157. doi: 10.1016/j.envsci.2013.02.002

Bull, I. (2018) Fruchtfolge erweitern oder Düngung reduzieren – Was bringt das für die N-Bilanz? Online: www.landwirtschaft-mv.de/Fachinformationen/Ackerbau/Winterraps/. Accessed: 13.01.2022

Bustamante, M., Robledo-Abad, C., Harper, R., Mbow, C., Ravindranat, N. H., Sperling, F., Haberl, H., Pinto, A. d. S., Smith, P. (2014) Co-benefits, trade-offs, barriers and policies for greenhouse gas mitigation in the agriculture, forestry and other land use (AFOLU) sector. Global change biology 20(10):3270-3290. doi: 10.1111/gcb.12591

Calegari, A., Hargrove, W. L., Rheinheimer, D. D. S., Ralisch, R., Tessier, D., Tourdonnet, S. de, Fatima Guimarães, M. de (2008) Impact of long-term no-tillage and cropping system management on soil organic carbon in an oxisol: a model for sustainability. Agronomy Journal 100(4):1013-1019. doi: 10.2134/agronj2007.0121

Camargo, G. G. T., Ryan, M. R., Richard, T. L. (2013) Energy use and greenhouse gas emissions from crop production using the Farm Energy Analysis Tool. BioScience 63(4):263-273. doi: 10.1525/bio.2013.63.4.6

Canalli, L. B. d. S., Da Costa, G. V., Volsi, B., Leocádio, A. L. M., Neves, C. S. V. J., Telles, T. S. (2020) Production and profitability of crop rotation systems in southern Brazil. Semina: Ciências Agrárias 41(6):2541. doi: 10.5433/1679-0359.2020v41n6p2541

Cannell, R. Q. (1985) Reduced tillage in north-west Europe—A review. Soil and Tillage Research 5(2):129-177. doi: 10.1016/0167-1987(85)90028-5

Cantarella, H., Otto, R., Soares, J. R., Silva, A. G. d. B. (2018) Agronomic efficiency of NBPT as a urease inhibitor: A review. Journal of Advanced Research 13:19-27. doi: 10.1016/j.jare.2018.05.008

Cantarella, H., Trivelin, P. C. O., Contin, T. L. M., Dias, F. L. F., Rossetto, R., Marcelino, R., Coimbra, R. B., Quaggio, J. A. (2008) Ammonia volatilisation from urease inhibitor-treated urea applied to sugarcane trash blankets. Scientia Agricola 65(4):397-401. doi: 10.1590/S0103-90162008000400011

Canter, C. E., Dunn, J. B., Han, J., Wang, Z., Wang, M. (2016) Policy implications of allocation methods in the life cycle analysis of integrated corn and corn stover ethanol production. BioEnergy Research 9(1):77-87. doi: 10.1007/s12155-015-9664-4

Causarano, H. J., Doraiswamy, P. C., McCarty, G. W., Hatfield, J. L., Milak, S., Stern, A. J. (2008) EPIC modeling of soil organic carbon sequestration in croplands of Iowa. Journal of Environmental Quality 37(4):1345-1353. doi: 10.2134/jeq2007.0277

Cerrato, M. E., Blackmer, A. M. (1990) Effects of nitrapyrin on corn yields and recovery of ammonium-n at 18 site-years in Iowa. Journal of Production Agriculture 3(4):513-521. doi: 10.2134/jpa1990.0513

Cerri, C. C., Bernoux, M., Maia, S. M. F., Cerri, C. E. P., Costa Junior, C., Feigl, B. J., Frazão, L. A., Mello, F. F. d. C., Galdos, M. V., Moreira, C. S., Carvalho, J. L. N. (2010) Greenhouse gas mitigation options in Brazil for land-use change, livestock and agriculture. Scientia Agricola 67(1):102-116. doi: 10.1590/S0103-90162010000100015

Chalise, K. S., Singh, S., Wegner, B. R., Kumar, S., Pérez-Gutiérrez, J. D., Osborne, S. L., Nleya, T., Guzman, J., Rohila, J. S. (2019) Cover crops and returning residue impact on soil organic carbon, bulk density, penetration resistance, water retention, infiltration, and soybean yield. Agronomy Journal 111(1):99-108. doi: 10.2134/agronj2018.03.0213

Chalk, P. M., Victoria, R. L., Muraoka, T., Piccolo, M. C. (1990) Effect of a nitrification inhibitor on immobilization and mineralization of soil and fertilizer nitrogen. Soil Biology and Biochemistry 22(4):533-538. doi: 10.1016/0038-0717(90)90190-B

Chapin, S. F., McFarland, J., David McGuire, A., Euskirchen, E. S., Ruess, R. W., Kielland, K. (2009) The changing global carbon cycle: linking plant-soil carbon dynamics to global consequences. Journal of Ecology 97(5):840-850. doi: 10.1111/j.1365-2745.2009.01529.x

Chen, R., Qin, Z., Han, J., Wang, M., Taheripour, F., Tyner, W., O'Connor, D., Duffield, J. (2018) Life cycle energy and greenhouse gas emission effects of biodiesel in the United States with induced land use change impacts. Bioresource technology 251:249-258. doi: 10.1016/j.biortech.2017.12.031

Chibanda, C., Agethen, K., Deblitz, C., Zimmer, Y., Almadani, M. I., Garming, H., Rohlmann, C., Schütte, J., Thobe, P., Verhaagh, M., Behrendt, L., Staub, D., Lasner, T. (2020) The Typical Farm Approach and Its Application by the Agri Benchmark Network. Agriculture 10(12):646. doi: 10.3390/agriculture10120646

Ciampitti, I. A., Salvagiotti, F. (2018) New insights into soybean biological nitrogen fixation. Agronomy Journal 110(4):1185-1196. doi: 10.2134/agronj2017.06.0348

Coggan, A., Whitten, S. M., Bennett, J. (2010) Influences of transaction costs in environmental policy. Ecological Economics 69(9):1777-1784. doi: 10.1016/j.ecolecon.2010.04.015

CONAB [Companhia Nacional de Abastecimento] (2021a) Historical data on harvests. Online: www.conab.gov.br/info-agro/safras/serie-historica-das-safras. Accessed: 02.09.2021

CONAB [Companhia Nacional de Abastecimento] (2021b) Tables of productions costs. Online: www.conab.gov.br/info-agro/custos-de-producao. Accessed: 02.09.2021

Conley, D. J., Paerl, H. W., Howarth, R. W., Boesch, D. F., Seitzinger, S. P., Havens, K. E., Lancelot, C., Likens, G. E. (2009) Ecology. Controlling eutrophication: nitrogen and phosphorus. Science 323(5917):1014-1015. doi: 10.1126/science.1167755

Córdova, S. C., Castellano, M. J., Dietzel, R., Licht, M. A., Togliatti, K., Martinez-Feria, R., Archontoulis, S. V. (2019) Soybean nitrogen fixation dynamics in Iowa, USA. Field Crops Research 236:165-176. doi: 10.1016/j.fcr.2019.03.018

Crawford, D. M., Chalk, P. M. (1992) Mineralization and immobilization of soil and fertilizer nitrogen with nitrification inhibitors and solvents. Soil Biology and Biochemistry 24(6):559-568. doi: 10.1016/0038-0717(92)90081-8

DeFries, R. S., Foley, J. A., Asner, G. P. (2004) Land-use choices: balancing human needs and ecosystem function. Frontiers in Ecology and the Environment 2(5):249-257. doi: 10.1890/1540-9295(2004)002[0249:LCBHNA]2.0.CO;2

Dehler, M. (2023) Maßnahmen zur Reduzierung des Pflanzenschutzmitteleinsatzes: Anpassungsoptionen, Kosten und Möglichkeiten zur umweltpolitischen Steuerung. Braunschweig: Johann-Heinrich-von-Thünen-Institut, 246 p. Thünen-Report 104

Dessart, F. J., Barreiro-Hurlé, J., van Bavel, R. (2019) Behavioural factors affecting the adoption of sustainable farming practices: a policy-oriented review. European Review of Agricultural Economics 46(3):417-471. doi: 10.1093/erae/jbz019

DESTATIS [Statistisches Bundesamt] (2010) Land- und Forstwirtschaft, Fischerei: Bodenbearbeitung, Bewässerung, Landschaftselemente Erhebung über landwirtschaftliche Produktionsmethoden (ELPM). Fachserie 3 Heft 5. Online: www.destatis.de/DE/Service/Bibliothek/_publikationen-fachserienliste-3.html. Accessed: 02.02.2022

DESTATIS [Statistisches Bundesamt] (2018) Wachstum und Ernte - Feldfrüchte - Fachserie 3 Reihe 3.2.1 - 16/2018. Online: www.destatis.de/DE/Home/_inhalt.html. Accessed: 11.10.2021

DESTATIS [Statistisches Bundesamt] (2020a) Land- und Forstwirtschaft, Fischerei: Bodennutzung der Betriebe (Struktur der Bodennutzung). Fachserie 3 Reihe 2.1.2. Online: www.statistischebibliothek.de/mir/receive/DEHeft_mods_00136944. Accessed: 18.01.2022

DESTATIS [Statistisches Bundesamt] (2020b) Wachstum und Ernte - Feldfrüchte - Fachserie 3 Reihe 3.2.1 - 16/2020. Online: www.destatis.de/DE/Home/_inhalt.html. Accessed: 11.10.2021

DESTATIS [Statistisches Bundesamt] (2022) Genesis-Online Databank. Online: www-genesis.destatis.de/genesis/online. Accessed: 12.01.2022

Deyn, G. B. de, Cornelissen, J. H. C., Bardgett, R. D. (2008) Plant functional traits and soil carbon sequestration in contrasting biomes. Ecology Letters 11(5):516-531. doi: 10.1111/j.1461-0248.2008.01164.x

Di, H. J., Cameron, K. C. (2002) Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies. Nutrient Cycling in Agroecosystems 64(3):237-256. doi: 10.1023/A:1021471531188

Dick, W. A., McCoy, E. L., Edwards, W. M., Lal, R. (1991) Continuous Application of No-Tillage to Ohio Soils. Agronomy Journal 83(1):65-73. doi: 10.2134/agronj1991.00021962008300010017x

Dooley, K., Christoff, P., Nicholas, K. A. (2018) Co-producing climate policy and negative emissions: trade-offs for sustainable land-use. Global Sustainability 1. doi: 10.1017/sus.2018.6

Dougherty, W. J., Collins, D., van Zwieten, L., Rowlings, D. W. (2016) Nitrification (DMPP) and urease (NBPT) inhibitors had no effect on pasture yield, nitrous oxide emissions, or nitrate leaching under irrigation in a hot-dry climate. Soil Research 54(5):675. doi: 10.1071/SR15330

Dozier, I. A., Behnke, G. D., Davis, A. S., Nafziger, E. D., Villamil, M. B. (2017) Tillage and cover cropping effects on soil properties and crop production in Illinois. Agronomy Journal 109(4):1261-1270. doi: 10.2134/agronj2016.10.0613

Drechsel, P., Heffer Patrick, Magen, H., Mikkelsen, R., Wichelns, D. (2015) Managing water and fertilizer for sustainable agricultural intensification, International Fertilizer Association (IFA); International Water Management Institute; International Potash Institute, 257 p

Drury, C. F., Yang, X., Reynolds, W. D., Calder, W., Oloya, T. O., Woodley, A. L. (2017) Combining urease and nitrification inhibitors with incorporation reduces ammonia and nitrous oxide emissions and increases corn yields. Journal of Environmental Quality 46(5):939-949. doi: 10.2134/jeq2017.03.0106

DWD [Deutscher Wetterdienst] (2022) Climate Data Center. Online: cdc.dwd.de/portal/. Accessed: 12.01.2022

Ehlers, M.-H., Huber, R., Finger, R. (2021) Agricultural policy in the era of digitalisation. Food Policy 100:102019. doi: 10.1016/j.foodpol.2020.102019

Eickhout, B., Bouwman, A. F., van Zeijts, H. (2006) The role of nitrogen in world food production and environmental sustainability. Agriculture, Ecosystems & Environment 116(1-2):4-14. doi: 10.1016/j.agee.2006.03.009

Elbakidze, L., McCarl, B. A. (2007) Sequestration offsets versus direct emission reductions: Consideration of environmental co-effects. Ecological Economics 60(3):564-571. doi: 10.1016/j.ecolecon.2005.12.014

EMATER [Empresa de Assistência Técnica e Extensão Rural do Estado Paraná] (2014) Evolution of direct seeding system in Paraná

EMBRAPA [Empresa Brasileira de Pesquisa Agropecuária] (2011) Importância da rotação de culturas para a produção agrícola sustentável no Paraná. Londrina: Embrapa Soja, 2011. Online: www.infoteca.cnptia.embrapa.br/infoteca/handle/doc/897259

EMBRAPA [Empresa Brasileira de Pesquisa Agropecuária] (2012) Atlas Climático da Regiao Sul do Brasil: Estados de Paraná, Santa Catarina e Rio Grande do Sul. Online: www.agricultura.rs.gov.br/upload/arquivos/202005/13110041-atlas-climatico-da-regiao-sul-do-brasil.pdf

Eory, V., Pellerin, S., Carmona Garcia, G., Lehtonen, H., Licite, I., Mattila, H., Lund-Sørensen, T., Muldowney, J., Popluga, D., Strandmark, L., Schulte, R. (2018) Marginal abatement cost curves for agricultural climate policy: State-of-the art, lessons learnt and future potential. Journal of Cleaner Production 182:705-716. doi: 10.1016/j.jclepro.2018.01.252

EPA [Environmental Protection Agency] (2010) Renewable Fuel Standard Program (RFS2) Regulatory Impact Analysis. Online: www.epa.gov/renewable-fuel-standard-program/renewable-fuel-standard-rfs2-final-rule-additional-resources

Espindula, M. C., Rocha, V. S., Souza, M. A. de, Campanharo, M., Pimentel, A. J. B. (2014) Urease inhibitor (NBPT) and efficiency of single or split application of urea in wheat crop. Revista Ceres 61(2):276-279. doi: 10.1590/S0034-737X2014000200016

EU [European Union] (2015) BioGrace: Harmonised Calculations of Bioenergy Greenhouse Emissions in Europe. Online: www.biograce.net/home. Accessed: 07.05.2021

FAO [Food and Agriculture Organization of the United Nations] (2022) FAOSTAT: Statistics database. Rome. Online: www.fao.org/faostat/en/#data. Accessed: 17.05.2018

Feliciano, D., Ledo, A., Hillier, J., Nayak, D. R. (2018) Which agroforestry options give the greatest soil and above ground carbon benefits in different world regions? Agriculture, Ecosystems & Environment 254:117-129. doi: 10.1016/j.agee.2017.11.032

Fellmann, T., Domínguez, I. P., Witzke, P., Weiss, F., Hristov, J., Barreiro-Hurle, J., Leip, A., Himics, M. (2021) Greenhouse gas mitigation technologies in agriculture: Regional circumstances and interactions determine cost-effectiveness. Journal of Cleaner Production 317:128406. doi: 10.1016/j.jclepro.2021.128406

Feuz, D. M., Skold, M. D. (1992) Typical farm theory in agricultural research. Journal of Sustainable Agriculture 2(2):43-58. doi: 10.1300/J064v02n02_05

Fischer, R. A. T., Edmeades, G. O. (2010) Breeding and Cereal Yield Progress. Crop Science 50:S-85-S-98. doi: 10.2135/cropsci2009.10.0564

Flaten, O., Koesling, M., Hansen, S., Veidal, A. (2019) Links between profitability, nitrogen surplus, greenhouse gas emissions, and energy intensity on organic and conventional dairy farms. Agroecology and Sustainable Food Systems 43(9):957-983. doi: 10.1080/21683565.2018.1544960

Follett, R. (2001) Soil management concepts and carbon sequestration in cropland soils. Soil and Tillage Research 61(1-2):77-92. doi: 10.1016/S0167-1987(01)00180-5

Follett, R. F., Shafer, S. R., Jawson, M. D., Franzluebbers, A. (2005) Research and implementation needs to mitigate greenhouse gas emissions from agriculture in the USA. Soil and Tillage Research 83(1):159-166. doi: 10.1016/j.still.2005.02.014

Fontoura, S. M. V., Bayer, C. (2009) Adubação nitrogenada para alto rendimento de milho em plantio direto na região centro-sul do Paraná. Revista Brasileira de Ciência do Solo 33(6):1721-1732. doi: 10.1590/s0100-06832009000600021

Fontoura, S. M. V., Bayer, C. (2010) Ammonia volatilization in no-till system in the south-central region of the State of Paraná, Brazil. Revista Brasileira de Ciência do Solo 34(5):1677-1684. doi: 10.1590/S0100-06832010000500020

Fraters, D., van Leeuwen, T., Boumans, L., Reijs, J. (2015) Use of long-term monitoring data to derive a relationship between nitrogen surplus and nitrate leaching for grassland and arable land on well-drained sandy soils in the Netherlands. Acta Agriculturae Scandinavica, Section B — Soil & Plant Science 65(sup2):144-154. doi: 10.1080/09064710.2014.956789

Frink, C. R., Waggoner, P. E., Ausubel, J. H. (1999) Nitrogen fertilizer: retrospect and prospect. Proceedings of the National Academy of Sciences of the United States of America 96(4):1175-1180. doi: 10.1073/pnas.96.4.1175

Fuentes, L. F. G., Souza, L. C. F. de, Serra, A. P., Rech, J., Vitorino, A. C. T. (2018) Corn agronomic traits and recovery of nitrogen from fertilizer during crop season and off-season. Pesquisa Agropecuária Brasileira 53(10):1158-1166. doi: 10.1590/S0100-204X2018001000009

Gandorfer, M., Meyer-Aurich, A. (2017) Economic potential of site-specific fertiliser application and harvest management. In: Precision Agriculture: Technology and Economic Perspectives. Springer, Cham: pp 79-92. Online: link.springer.com/chapter/10.1007/978-3-319-68715-5_3

Garcia, P. L., González-Villalba, H. A., Sermarini, R. A., Trivelin, P. C. O. (2018a) Nitrogen use efficiency and nutrient partitioning in maize as affected by blends of controlled-release and conventional urea. Archives of Agronomy and Soil Science 64(14):1944-1962. doi: 10.1080/03650340.2018.1469746

Garcia, R. A., Ceccon, G., Da Sutier, G. A. S., Santos, A. L. F. d. (2018b) Soybean-corn succession according to seeding date. Pesquisa Agropecuária Brasileira 53(1):22-29. doi: 10.1590/s0100-204x2018000100003

Gardner, T. A., Burgess, N. D., Aguilar-Amuchastegui, N., Barlow, J., Berenguer, E., Clements, T., Danielsen, F., Ferreira, J., Foden, W., Kapos, V., Khan, S. M., Lees, A. C., Parry, L., Roman-Cuesta, R. M., Schmitt, C. B., Strange, N., Theilade, I., Vieira, I. C. (2012) A framework for integrating biodiversity concerns into national REDD+ programmes. Biological Conservation 154:61-71. doi: 10.1016/j.biocon.2011.11.018

Gebhardt, M. R., Daniel, T. C., Schweizer, E. E., Allmaras, R. R. (1985) Conservation tillage. Science 230(4726):625-630. doi: 10.1126/science.230.4726.625

Geospatial Laboratory for Soil Informatics (2019) Iowa Soil Drainage Class. Bohn, Meyer; McDanel, Joshua; Miller; Bradley. Online: glsi.agron.iastate.edu/2019/07/22/download-iowa-soil-drainage-class-maps-gssurgo/

Giacomini, S. J., Aita, C., Chiapinotto, I. C., Hübner, A. P., Marques, M. G., Cadore, F. (2004) Consorciação de plantas de cobertura antecedendo o milho em plantio direto: II - Nitrogênio acumulado pelo milho e produtividade de grãos. Revista Brasileira de Ciência do Solo 28(4):751-762. doi: 10.1590/S0100-06832004000400015

Giacomini, S. J., Aita, C., Vendruscolo, E. R. O., Cubilla, M., Nicoloso, R. S., Fries, M. R. (2003) Matéria seca, relação C/N e acúmulo de nitrogênio, fósforo e potássio em misturas de plantas de cobertura de solo. Revista Brasileira de Ciência do Solo 27(2):325-334. doi: 10.1590/s0100-06832003000200012

Gioacchini, P., Nastri, A., Marzadori, C., Giovannini, C., Vittori Antisari, L., Gessa, C. (2002) Influence of urease and nitrification inhibitors on N losses from soils fertilized with urea. Biology and Fertility of Soils 36(2):129-135. doi: 10.1007/s00374-002-0521-1

Gnansounou, E., Panichelli, L., Dauriat, A., Villegas, J. D. (2008) Accounting for indirect land-use changes in GHG balances of biofuels: Review of current approaches. Online: infoscience.epfl.ch/record/121496

Grandy, A. S., Robertson, G. P. (2007) Land-use intensity effects on soil organic carbon accumulation rates and mechanisms. Ecosystems 10(1):59-74. doi: 10.1007/s10021-006-9010-y

Grant, R. F., Pattey, E., Goddard, T. W., Kryzanowski, L. M., Puurveen, H. (2006) Modeling the effects of fertilizer application rate on nitrous oxide emissions. Soil Science Society of America Journal 70(1):235-248. doi: 10.2136/sssaj2005.0104

Grodzki, L., Caramori, P. H., Bootsma, A., Oliveira, D. de, Gomes, J. (1996) Riscos de ocorrência de geada no Estado do Paraná. Revista Brasileira de Agrometeorologia, Santa Maria 4(1):93-99

Gruber, S., Pekrun, C., Möhring, J., Claupein, W. (2012) Long-term yield and weed response to conservation and stubble tillage in SW Germany. Soil and Tillage Research 121:49-56. doi: 10.1016/j.still.2012.01.015

Gu, C., Riley, W. J. (2010) Combined effects of short term rainfall patterns and soil texture on soil nitrogen cycling - a modeling analysis. Journal of Contaminant Hydrology 112(1-4):141-154. doi: 10.1016/j.jconhyd.2009.12.003

Guertal, E. A. (2009) Slow-release nitrogen fertilizers in vegetable production: a review. HortTechnology 19(1):16-19. doi: 10.21273/HORTSCI.19.1.16

Haddaway, N. R., Hedlund, K., Jackson, L. E., Kätterer, T., Lugato, E., Thomsen, I. K., Jørgensen, H. B., Isberg, P.-E. (2017) How does tillage intensity affect soil organic carbon? A systematic review. Environmental Evidence 6(1):77. doi: 10.1186/s13750-017-0108-9

Halvorson, A. D., Snyder, C. S., Blaylock, A. D., Del Grosso, S. J. (2014) Enhanced-Efficiency Nitrogen Fertilizers: Potential Role in Nitrous Oxide Emission Mitigation. Agronomy Journal 106(2):715-722. doi: 10.2134/agronj2013.0081

Hashemi, M., Farsad, A., Sadeghpour, A., Weis, S. A., Herbert, S. J. (2013) Cover-crop seeding-date influence on fall nitrogen recovery. Journal of Plant Nutrition and Soil Science 176(1):69-75. doi: 10.1002/jpln.201200062

Hasler, K., Bröring, S., Omta, S., Olfs, H.-W. (2015) Life cycle assessment (LCA) of different fertilizer product types. European Journal of Agronomy 69:41-51. doi: 10.1016/j.eja.2015.06.001

Hemme, T., Isermeyer, F., Deblitz, C. (1997) TIPI-CAL Version 1.0: ein Modell zur Politik- und Technikfolgenabschätzung für typische Betriebe im internationalen Vergleich, Bundesforschungsanstalt für Landwirtschaft

Hively, W. D., Lang, M., McCarty, G. W., Keppler, J., Sadeghi, A., McConnell, L. L. (2009) Using satellite remote sensing to estimate winter cover crop nutrient uptake efficiency. Journal of Soil and Water Conservation 64(5):303-313. doi: 10.2489/jswc.64.5.303

Hofstrand, D. (2020a) Tracking the Profitability of Corn Production. Online: www.extension.iastate.edu/agdm/crops/html/a1-85.html. Accessed: 06.08.2020

Hofstrand, D. (2020b) Tracking the Profitability of Soybean Production. Online: www.extension.iastate.edu/agdm/crops/html/a1-86.html. Accessed: 06.08.2020

Holland, J. M. (2004) The environmental consequences of adopting conservation tillage in Europe: reviewing the evidence. Agriculture, Ecosystems & Environment 103(1):1-25. doi: 10.1016/j.agee.2003.12.018

Holtsmark, B. (2015) Quantifying the global warming potential of CO 2 emissions from wood fuels. GCB Bioenergy 7(2):195-206. doi: 10.1111/gcbb.12110

Hoxha, A., Christensen, B. (2018) The carbon footprint of fertiliser production: Regional reference values. [Colchester]: International Fertiliser Society, 1 volume. Proceedings 805

Huang, J., Gómez-Dans, J. L., Huang, H., Ma, H., Wu, Q., Lewis, P. E., Liang, S., Chen, Z., Xue, J.-H., Wu, Y., Zhao, F., Wang, J., Xie, X. (2019) Assimilation of remote sensing into crop growth models: Current status and perspectives. Agricultural and Forest Meteorology 276-277:107609. doi: 10.1016/j.agrformet.2019.06.008

Huang, J., Xu, C., Ridoutt, B. G., Wang, X., Ren, P. (2017) Nitrogen and phosphorus losses and eutrophication potential associated with fertilizer application to cropland in China. Journal of Cleaner Production 159:171-179. doi: 10.1016/j.jclepro.2017.05.008

Huggins, D. R., Clapp, C. E., Allmaras, R. R., Lamb, J. A., Layese, M. F. (1998) Carbon dynamics in corn-soybean sequences as estimated from natural carbon-13 abundance. Soil Science Society of America Journal 62(1):195-203. doi: 10.2136/sssaj1998.03615995006200010026x

Hunter, M. C., Schipanski, M. E., Burgess, M. H., LaChance, J. C., Bradley, B. A., Barbercheck, M. E., Kaye, J. P., Mortensen, D. A. (2019) Cover crop mixture effects on maize, soybean, and wheat yield in rotation. Agricultural & Environmental Letters 4(1):180051. doi: 10.2134/ael2018.10.0051

Hussain, S., Hussain, S., Guo, R., Sarwar, M., Ren, X., Krstic, D., Aslam, Z., Zulifqar, U., Rauf, A., Hano, C., El-Esawi, M. A. (2021) Carbon sequestration to avoid soil degradation: a review on the role of conservation tillage. Plants 10(10):2001. doi: 10.3390/plants10102001

Huynh, H. T., Hufnagel, J., Wurbs, A., Bellingrath-Kimura, S. D. (2019) Influences of soil tillage, irrigation and crop rotation on maize biomass yield in a 9-year field study in Müncheberg, Germany. Field Crops Research 241:107565. doi: 10.1016/j.fcr.2019.107565

Hyatt, J., Wendroth, O., Egli, D. B., TeKrony, D. M. (2007) Soil compaction and soybean seedling emergence. Crop Science 47(6):2495-2503. doi: 10.2135/cropsci2007.03.0171

IBGE [Brasilian Institute of Geography and Statistics] (2018) Censo Agropecuário 2017. Online: sidra.ibge.gov.br/pesquisa/censo-agropecuario/censo-agropecuario-2017

ICF (2013) Greenhouse Gas Mitigation Options and Costs for Agricultural Land and Animal Production within the United States. Online: www.usda.gov/sites/default/files/documents/GHG_Mitigation_Options.pdf

IFA [International Fertilizer Association] (2001) Global estimates of gaseous emissions of NH 3, NO and N2 O from agricultural land, 1. version. Paris, 106 p

IFA [International Fertilizer Association] (2020) N, P2O5 and K2O removal by crops. Online: api.ifastat.org/reports/download/12965

Iowa State University (2016) Estimated Costs of Crop Production in Iowa - 2016. Online: www.extension.iastate.edu/agdm/crops/pdf/a1-20_2016.pdf. Accessed: 26.08.2020

Iowa State University (2017) Estimated Costs of Crop Production in Iowa - 2017. Online: www.extension.iastate.edu/agdm/crops/pdf/a1-20_2017.pdf. Accessed: 26.08.2020

Iowa State University (2018) Estimated Costs of Crop Production in Iowa - 2018. Online: www.extension.iastate.edu/agdm/crops/pdf/a1-20_2018.pdf. Accessed: 26.08.2020

Iowa State University (2020) Last and First Frost Dates in Iowa | Horticulture and Home Pest News. Online: hortnews.extension.iastate.edu/2005/5-4-2005/iowafrostdates.html. Accessed: 30.07.2020

IPCC [Intergovernmental Panel on Climate Change] (ed) (2006) 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Volume 4, Chapter 11: N2O emissions from managed soils, and CO2 emissions from lime and urea application, 54 p

IPCC [Intergovernmental Panel on Climate Change] (2007) Climate Change 2007: The Physical Science Basis.: Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, MA: Cambridge University Press

IPCC [Intergovernmental Panel on Climate Change] (2013) Climate Change 2013: The Physical Science Basis.: Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, MA: Cambridge University Press

IPCC [Intergovernmental Panel on Climate Change] (2014) Climate Change 2014: Mitigation of Climate Change - Agriculture, Forestry and Other Land Use (AFOLU): Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, MA: Cambridge University Press

IPCC [Intergovernmental Panel on Climate Change] (ed) (2019a) 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Volume 4: Agricultre, Forestry and Other Land use. Japan: IGES

IPCC [Intergovernmental Panel on Climate Change] (2019b) IPCC Special Report on Climate Change, Desertification, Land Degradation, Sustainable Land Management, Food Security, and Greenhouse gas fluxes in Terrestrial Ecosystems, 43 p

Jacobs, A., Flessa, H., Don, A. (2018) Landwirtschaftlich genutzte Böden in Deutschland: Ergebnisse der Bodenzustandserhebung. Braunschweig, Germany: Johann Heinrich von Thünen-Institut. Thünen Report 64

Jacobs, A., Poeplau, C., Weiser, C., Fahrion-Nitschke, A., Don, A. (2020) Exports and inputs of organic carbon on agricultural soils in Germany. Fertilizer research 118(3):249-271. doi: 10.1007/s10705-020-10087-5

Jayaraman, S., Bandyopadhyay, K. K., Naorem, A. K., Sinha, N. K., Mohanty, M., Hati, K. M., Patra, A. K., Chaudhari, S. K., Dalal, R. C., Lal, R. (2021) Soil Carbon Sequestration Through Conservation Tillage and Residue Management. In: Jayaraman S, Dalal RC, Patra AK, Chaudhari SK (eds) Conservation Agriculture: A Sustainable Approach for Soil Health and Food Security: Conservation Agriculture for Sustainable Agriculture, 1st ed. 2021. Singapore: Springer Singapore; Imprint Springer: pp 299-319

Jayathilake, H. M., Prescott, G. W., Carrasco, L. R., Rao, M., Symes, W. S. (2021) Drivers of deforestation and degradation for 28 tropical conservation landscapes. Ambio 50(1):215-228. doi: 10.1007/s13280-020-01325-9

Jian, J., Du, X., Reiter, M. S., Stewart, R. D. (2020) A meta-analysis of global cropland soil carbon changes due to cover cropping. Soil Biology and Biochemistry 143:107735. doi: 10.1016/j.soilbio.2020.107735

Jie, C., Jing-zhang, C., Man-zhi, T., Zi-tong, G. (2002) Soil degradation: a global problem endangering sustainable development. Journal of Geographical Sciences 12(2):243-252. doi: 10.1007/BF02837480

Johnson, J. M.-F., Franzluebbers, A. J., Weyers, S. L., Reicosky, D. C. (2007) Agricultural opportunities to mitigate greenhouse gas emissions. Environmental pollution (Barking, Essex : 1987) 150(1):107-124. doi: 10.1016/j.envpol.2007.06.030

Jones, R. R., Weyer, P. J., DellaValle, C. T., Inoue-Choi, M., Anderson, K. E., Cantor, K. P., Krasner, S., Robien, K., Freeman, L. E. B., Silverman, D. T., Ward, M. H. (2016) Nitrate from drinking water and diet and bladder cancer among postmenopausal women in Iowa. Environmental health perspectives 124(11):1751-1758. doi: 10.1289/EHP191

JRC [European Joint Research Center] (2022) LULUCF - IPCC Classifications. Online: forest.jrc.ec.europa.eu/en/activities/lulucf/ipcc-classifications/. Accessed: 25.01.2022

Kahiluoto, H., Rimhanen, K., Rötter, R., Tseganeh, B. (2012) Mitigation of climate change to enhance food security: an analytical framework. Forum for Development Studies 39(1):51-73. doi: 10.1080/08039410.2011.635381

Karatay, Y. N., Meyer-Aurich, A., Hunstock, U., Stella, T., Nendel, C. (2019) Kosteneffiziente Vermeidung von Treibhausgasemissionen durch Düngereduktion: standortspezifische Analyse mit simulierten

Ertragsfunktionen. In: Meyer-Aurich A, Gandorfer M, Barta N, Gronauer A, Kantelhardt J, Floto H (eds) 39. GIL-Jahrestagung, Digitalisierung für landwirtschaftliche Betriebe in kleinstrukturierten Regionen - ein Widerspruch in sich? Bonn: Gesellschaft für Informatik e.V: pp 107-112

Kasampalis, D., Alexandridis, T., Deva, C., Challinor, A., Moshou, D., Zalidis, G. (2018) Contribution of remote sensing on crop models: a review. Journal of Imaging 4(4):52. doi: 10.3390/jimaging4040052

Kaspar, T. C., Parkin, T. B., Jaynes, D. B., Cambardella, C. A., Meek, D. W., Jung, Y. S. (2006) Examining changes in soil organic carbon with oat and rye cover crops using terrain covariates. Soil Science Society of America Journal 70(4):1168-1177. doi: 10.2136/sssaj2005.0095

Kathrin, H., Stefanie, B., Onno SWF, O., Hans-Werner, O. (2017) Eco-innovations in the German fertilizer supply chain: Impact on the carbon footprint of fertilizers. Plant, Soil and Environment 63(No. 12):531-544. doi: 10.17221/499/2017-PSE

Kesicki, F., Ekins, P. (2012) Marginal abatement cost curves: a call for caution. Climate Policy 12(2):219-236. doi: 10.1080/14693062.2011.582347

Kesicki, F., Strachan, N. (2011) Marginal abatement cost (MAC) curves: confronting theory and practice. Environmental Science & Policy 14(8):1195-1204. doi: 10.1016/j.envsci.2011.08.004

Klocke, N. L., Watts, D. G., Schneekloth, J. P., Davison, D. R., Todd, R. W., Parkhurst, A. M. (1999) NITRATE LEACHING IN IRRIGATED CORN AND SOYBEAN IN A SEMI-ARID CLIMATE. Transactions of the ASAE 42(6):1621-1630. doi: 10.13031/2013.13328

Komainda, M., Taube, F., Kluß, C., Herrmann, A. (2018) Effects of catch crops on silage maize (Zea mays L.): yield, nitrogen uptake efficiency and losses. Fertilizer research 110(1):51-69. doi: 10.1007/s10705-017-9839-9

Komatsuzaki, M., Ohta, H. (2007) Soil management practices for sustainable agro-ecosystems. Sustainability Science 2(1):103-120. doi: 10.1007/s11625-006-0014-5

Kopittke, P. M., Menzies, N. W., Wang, P., McKenna, B. A., Lombi, E. (2019) Soil and the intensification of agriculture for global food security. Environment international 132:105078. doi: 10.1016/j.envint.2019.105078

Kowarsch, M., Garard, J., Riousset, P., Lenzi, D., Dorsch, M. J., Knopf, B., Harrs, J.-A., Edenhofer, O. (2016) Scientific assessments to facilitate deliberative policy learning. Palgrave Communications 2(1). doi: 10.1057/palcomms.2016.92

Kraiser, T., Gras, D. E., Gutiérrez, A. G., González, B., Gutiérrez, R. A. (2011) A holistic view of nitrogen acquisition in plants. Journal of Experimental Botany 62(4):1455-1466. doi: 10.1093/jxb/erq425

Krug, J. (2013) Perspektiven ackerbaulicher Grenzstandorte in Nordostdeutschland: Übertragbarkeit extensiver Produktionssysteme überseeischer Trockenstandorte. Braunschweig, Thünen Report. Online: www.thuenen.de/media/publikationen/thuenen-report/Thuenen_Report_06.pdf

Kühn, T. (2018) Gruppendiskussionen: Ein Praxis-Handbuch, 2. Aufl. 2018. Wiesbaden: Springer VS, 331 p. SpringerLink Bücher

Kuo, S., Sainju, U. M. (1998) Nitrogen mineralization and availability of mixed leguminous and non-leguminous cover crop residues in soil. Biology and Fertility of Soils 26(4):346-353. doi: 10.1007/s003740050387

Lal, R. (2004a) Soil carbon sequestration impacts on global climate change and food security. Science 304(5677):1623-1627. doi: 10.1126/science.1097396

Lal, R. (2004b) Soil carbon sequestration to mitigate climate change. Geoderma 123(1-2):1-22. doi: 10.1016/j.geoderma.2004.01.032

Lal, R. (2010) Beyond Copenhagen: mitigating climate change and achieving food security through soil carbon sequestration. Food Security 2(2):169-177. doi: 10.1007/s12571-010-0060-9

Lal, R. (2014) Societal value of soil carbon. Journal of Soil and Water Conservation 69(6):186A-192A. doi: 10.2489/jswc.69.6.186A

Lal, R., Reicosky, D. C., Hanson, J. D. (2007) Evolution of the plow over 10,000 years and the rationale for no-till farming. Soil and Tillage Research 93(1):1-12. doi: 10.1016/j.still.2006.11.004

Lamb, W. F., Wiedmann, T., Pongratz, J., Andrew, R., Crippa, M., Olivier, J. G. J., Wiedenhofer, D., Mattioli, G., Khourdajie, A. A., House, J., Pachauri, S., Figueroa, M., Saheb, Y., Slade, R., Hubacek, K., Sun, L., Ribeiro, S. K., Khennas, S., La Rue Can, S. de, Chapungu, L., Davis, S. J., Bashmakov, I., Dai, H., Dhakal, S., Tan, X., Geng, Y., Gu, B., Minx, J. (2021) A review of trends and drivers of greenhouse gas emissions by sector from 1990 to 2018. Environmental Research Letters 16(7):73005. doi: 10.1088/1748-9326/abee4e

Lankoski, J., Lötjönen, S., Ollikainen, M. (2020) Climate change mitigation and agriculture: measures, costs and policies – A literature review. Agricultural and Food Science 29(2). doi: 10.23986/afsci.85830

Lassaletta, L., Billen, G., Grizzetti, B., Anglade, J., Garnier, J. (2014) 50 year trends in nitrogen use efficiency of world cropping systems: the relationship between yield and nitrogen input to cropland. Environmental Research Letters 9(10):105011. doi: 10.1088/1748-9326/9/10/105011

Lavoie, G., Gunjal, K., Raghavan, G. S. V. (1991) Soil compaction, machinery selection, and optimum crop planning. Transactions of the ASAE 34(1):2. doi: 10.13031/2013.31614

Lawson, A., Fortuna, A. M., Cogger, C., Bary, A., Stubbs, T. (2013) Nitrogen contribution of rye–hairy vetch cover crop mixtures to organically grown sweet corn. Renewable Agriculture and Food Systems 28(1):59-69. doi: 10.1017/S1742170512000014

Leahy, S., Clark, H., Reisinger, A. (2020) Challenges and prospects for agricultural greenhouse gas mitigation pathways consistent with the Paris Agreement. Frontiers in sustainable food systems 4. doi: 10.3389/fsufs.2020.00069

Lee, M., Lee, J., Lee, J., KIM, Y., PARK, Y., Lee, K. (2017) Uncertainty analysis of a GHG emission model output using the block bootstrap and Monte Carlo Simulation. Sustainability 9(9):1522. doi: 10.3390/su9091522

LFA [Landesforschungsanstalt für Landwirtschaft und Fischerei Mecklenburg-Vorpommern] (2014a) Ergebnisse zur N-Düngung von Qualitätsweizen im Nordosten Deutschlands, hg. v. Ines Bull, Landesforschungsanstalt für Landwirtschaft und Fischerei Mecklenburg-Vorpommern (LFA). Online: www.wrrl-mv-landwirtschaft.de/node/18

LFA [Landesforschungsanstalt für Landwirtschaft und Fischerei Mecklenburg-Vorpommern] (2014b) Stickstoffkonservierung durch Zwischenfrüchte über Winter in Abhängigkeit von der Pflanzenart und dem Aussaattermin, hg. v. Ines Bull. Online: www.wrrl-mv-landwirtschaft.de/node/5

LFA [Landesforschungsanstalt für Landwirtschaft und Fischerei Mecklenburg-Vorpommern] (2019a) Mähdruschfruchtauswertung 2018. Online: www.landwirtschaft-mv.de/Fachinformationen/Agraroekonomie

LFA [Landesforschungsanstalt für Landwirtschaft und Fischerei Mecklenburg-Vorpommern] (2019b) Untersuchungen zu acker- und pflanzenbaulichen Wirkungen und zur Wirtschaftlichkeit von Fruchtfolgen in Marktfruchtbaubetrieben Mecklenburg-Vorpommerns: Erste Bewertung betrieblicher Fruchtfolgen. Online: www.landwirtschaft-mv.de/Fachinformationen/Agraroekonomie

LFA [Landesforschungsanstalt für Landwirtschaft und Fischerei Mecklenburg-Vorpommern] (2020) Mähdruschfruchtauswertung 2019. Online: www.landwirtschaft-mv.de/Fachinformationen/Agraroekonomie

LFA [Landesforschungsanstalt für Landwirtschaft und Fischerei Mecklenburg-Vorpommern] (2021) Mähdruschfruchtauswertung 2020. Online: www.landwirtschaft-mv.de/Fachinformationen/Agraroekonomie

LfL [Bayerische Landesanstalt für Landwirtschaft] (2001) Zehn Jahre Stickstoff-Monitoring in Bayern. Online: www.lfl.bayern.de/iab/duengung/032228/index.php

Li, S., Wang, Z., Miao, Y., Li, S. (2014) Soil organic nitrogen and its contribution to crop production. Journal of Integrative Agriculture 13(10):2061-2080. doi: 10.1016/S2095-3119(14)60847-9

Li, T., Zhang, W., Yin, J., Chadwick, D., Norse, D., Lu, Y., Liu, X., Chen, X., Zhang, F., Powlson, D., Dou, Z. (2018) Enhanced-efficiency fertilizers are not a panacea for resolving the nitrogen problem. Global change biology 24(2):e511-e521. doi: 10.1111/gcb.13918

Li, Z., Zeng, Z., Tian, D., Wang, J., Fu, Z., Zhang, F., Zhang, R., Chen, W., Luo, Y., Niu, S. (2020) Global patterns and controlling factors of soil nitrification rate. Global change biology 26(7):4147-4157. doi: 10.1111/gcb.15119

Licht, M. A., Al-Kaisi, M. M. (2005) Strip-tillage effect on seedbed soil temperature and other soil physical properties. Soil and Tillage Research 80(1-2):233-249. doi: 10.1016/j.still.2004.03.017

Liu, H., Zak, D., Rezanezhad, F., Lennartz, B. (2019) Soil degradation determines release of nitrous oxide and dissolved organic carbon from peatlands. Environmental Research Letters 14(9):94009. doi: 10.1088/1748-9326/ab3947

Liu, Y., Swinton, S. M., Miller, N. R. (2006) Is site-specific yield response consistent over time? Does it pay? American Journal of Agricultural Economics 88(2):471-483. doi: 10.1111/j.1467-8276.2006.00872.x

Liu, Y., Wu, L., Baddeley, J. A., Watson, C. A. (2011) Models of biological nitrogen fixation of legumes. A review. Agronomy for Sustainable Development 31(1):155-172. doi: 10.1051/agro/2010008

Lotter, D. W., Seidel, R., Liebhardt, W. (2003) The performance of organic and conventional cropping systems in an extreme climate year. American Journal of Alternative Agriculture 18(3):146-154. doi: 10.1079/AJAA200345

Luo, Z., Wang, E., Sun, O. J. (2010) Can no-tillage stimulate carbon sequestration in agricultural soils? A meta-analysis of paired experiments. Agriculture, Ecosystems & Environment 139(1-2):224-231. doi: 10.1016/j.agee.2010.08.006

MacLeod, M., Eory, V., Gruère, G., Lankoski, J. (2015) Cost-Effectiveness of Greenhouse Gas Mitigation Measures for Agriculture - OECD Food, Agriculture and Fisheries Papers No. 89

MAGYP [Ministry of Agriculture, Livestock and Fisheries] (2021) Agricultural Estimations. Online: datosestimaciones.magyp.gob.ar/

Maheswari, M., Murthy, A., Shanker, A. K. (2017) Nitrogen nutrition in crops and its importance in crop quality. In: The Indian Nitrogen Assessment. Elsevier: pp 175-186

Mani, T., Murugan, P., Mahinpey, N. (2011) Pyrolysis of oat straw and the comparison of the product yield to wheat and flax straw pyrolysis. Energy & Fuels 25(7):2803-2807. doi: 10.1021/ef200546v

Martre, P., Jamieson, P. D., Semenov, M. A., Zyskowski, R. F., Porter, J. R., Triboi, E. (2006) Modelling protein content and composition in relation to crop nitrogen dynamics for wheat. European Journal of Agronomy 25(2):138-154. doi: 10.1016/j.eja.2006.04.007

Masnadi, M. S., El-Houjeiri, H. M., Schunack, D., Li, Y., Englander, J. G., Badahdah, A., Monfort, J.-C., Anderson, J. E., Wallington, T. J., Bergerson, J. A., Gordon, D., Koomey, J., Przesmitzki, S., Azevedo, I. L., Bi, X. T., Duffy, J. E., Heath, G. A., Keoleian, G. A., McGlade, C., Meehan, D. N., Yeh, S., You, F., Wang, M., Brandt, A. R. (2018) Global carbon intensity of crude oil production. Science 361(6405):851-853. doi: 10.1126/science.aar6859

Mazzoncini, M., Sapkota, T. B., Bàrberi, P., Antichi, D., Risaliti, R. (2011) Long-term effect of tillage, nitrogen fertilization and cover crops on soil organic carbon and total nitrogen content. Soil and Tillage Research 114(2):165-174. doi: 10.1016/j.still.2011.05.001

Mbow, C., Skole, D., Dieng, M., Justice, C, Kwesha, D, Mane, L., El Gamri, M., von Vordzogbe, V, Virji, H (2012) Challenges and prospects for REDD+ in Africa: desk review of REDD+ implementation in Africa. Online: www.worldagroforestry.org/publication/challenges-and-prospects-redd-africa-desk-review-redd-implementation-africa

McAllister, C. H., Beatty, P. H., Good, A. G. (2012) Engineering nitrogen use efficient crop plants: the current status. Plant Biotechnology Journal 10(9):1011-1025. doi: 10.1111/j.1467-7652.2012.00700.x

McCann, L. (2013) Transaction costs and environmental policy design. Ecological Economics 88:253-262. doi: 10.1016/j.ecolecon.2012.12.012

McCarl, B. A., Yu, C.-H., Attavanich, W. (2021) Climate change impacts and strategies for mitigation and adaptation in agriculture. Atmosphere 12(5):545. doi: 10.3390/atmos12050545

McNunn, G., Karlen, D. L., Salas, W., Rice, C. W., Mueller, S., Muth, D., Seale, J. W. (2020) Climate smart agriculture opportunities for mitigating soil greenhouse gas emissions across the U.S. Corn-Belt. Journal of Cleaner Production 268:122240. doi: 10.1016/j.jclepro.2020.122240

Meade, C. V., Mullins, E. D. (2005) GM crop cultivation in Ireland: ecological and economic considerations. Biology and Environment: Proceedings of the Royal Irish Academy 105B(1):33-52

Meisinger (1991) Cover crops for clean water. Proceedings of an international conference, 57 p

Melillo, J. M. (2021) Disruption of the global nitrogen cycle: A grand challenge for the twenty-first century : This article belongs to Ambio's 50th Anniversary Collection. Theme: Eutrophication. Ambio 50(4):759-763. doi: 10.1007/s13280-020-01429-2

Michel, M., Vietinghoff, J., Michel, V. (2007) Produktionsfuntionen und N-Bilanzen bei Winterweizen, Landesforschungsanstalt für Landwirtschaft und Fischerei Mecklenburg-Vorpommern (LFA). Beiträge zum Sorten- und Versuchswesen und zur Biostatistik - Heft 37. Online: www.landwirtschaft-mv.de/

Millar, N., Robertson, G. P., Grace, P. R., Gehl, R. J., Hoben, J. P. (2010) Nitrogen fertilizer management for nitrous oxide (N2O) mitigation in intensive corn (Maize) production: an emissions reduction protocol for US Midwest agriculture. Mitigation and Adaptation Strategies for Global Change 15(2):185-204. doi: 10.1007/s11027-010-9212-7

Modolo, L. V., da-Silva, C. J., Brandão, D. S., Chaves, I. S. (2018) A minireview on what we have learned about urease inhibitors of agricultural interest since mid-2000s. Journal of Advanced Research 13:29-37. doi: 10.1016/j.jare.2018.04.001

Moore, E. B., Wiedenhoeft, M. H., Kaspar, T. C., Cambardella, C. A. (2014) Rye cover crop effects on soil quality in no-till corn silage-soybean cropping systems. Soil Science Society of America Journal 78(3):968-976. doi: 10.2136/sssaj2013.09.0401

Moraes Sá, J. C. de, Tivet, F., Lal, R., Briedis, C., Hartman, D. C., dos Santos, J. Z., dos Santos, J. B. (2014) Long-term tillage systems impacts on soil C dynamics, soil resilience and agronomic productivity of a Brazilian Oxisol. Soil and Tillage Research 136:38-50. doi: 10.1016/j.still.2013.09.010

Moran, D., Lucas, A., Barnes, A. (2013) Mitigation win–win. Nature Climate Change 3(7):611-613. doi: 10.1038/nclimate1922

Morland, J. K., Feagin, J. R., Orum, A. M., Sjoberg, G. (1992) A Case for the case study. Social Forces 71(1):240. doi: 10.2307/2579984

Morris, T. F., Murrell, T. S., Beegle, D. B., Camberato, J. J., Ferguson, R. B., Grove, J., Ketterings, Q., Kyveryga, P. M., Laboski, C. A., McGrath, J. M., Meisinger, J. J., Melkonian, J., Moebius-Clune, B. N., Nafziger, E. D., Osmond, D., Sawyer, J. E., Scharf, P. C., Smith, W., Spargo, J. T., Es, H. M., Yang, H. (2018) Strengths and limitations of nitrogen rate recommendations for corn and opportunities for improvement. Agronomy Journal 110(1):1-37. doi: 10.2134/agronj2017.02.0112

Mosier, A. (2004) Agriculture and the Nitrogen Cycle: Assessing the Impacts of Fertilizer Use on Food Production and the Environment. Washington: Island Press, 320 p. Scientific Committee on Problems of the Environment (SCOPE) Ser v.65

Mosier, A., Syers, J. K., Freney, J. R. (2013) Agriculture and the Nitrogen Cycle: Assessing the Impacts of Fertilizer Use on Food Production and the Environment. Island Press

Mußhoff, O., Hirschauer, N. (2020) Modernes Agrarmanagement: Betriebswirtschaftliche Analyse- und Planungsverfahren, 5., überarbeitete und erweiterte Auflage. München: Verlag Franz Vahlen. Online: ebookcentral.proquest.com/lib/kxp/detail.action?docID=6181168

Natural Resources Conservation Service (2021) Distribution Maps of Dominant Soil Orders. Online: www.nrcs.usda.gov/wps/portal/nrcs/. Accessed: 10.03.2021

Nehring, K. (2011) Farm level implications of high commodity prices: An assessment of adaption strategies and potentials in selected regions in Australia and Germany. Zugl.: Göttingen, Univ., Diss. 2011. Braunschweig: vTI, 93 p. Landbauforschung Sonderheft 349

Nicoloso, R. S., Rice, C. W. (2021) Intensification of no-till agricultural systems: An opportunity for carbon sequestration. Soil Science Society of America Journal 85(5):1395-1409. doi: 10.1002/saj2.20260

NOAA [National Oceanic and Atmospheric Administration] (2020) Applied Climate Information System. Online: www.weather.gov/. Accessed: 30.07.2020

Noble, B. F., Christmas, L. M. (2008) Strategic environmental assessment of greenhouse gas mitigation options in the Canadian agricultural sector. Environmental management 41(1):64-78. doi: 10.1007/s00267-007-9017-y

O.Nyumba, T., Wilson, K., Derrick, C. J., Mukherjee, N. (2018) The use of focus group discussion methodology: Insights from two decades of application in conservation. Methods in Ecology and Evolution 9(1):20-32. doi: 10.1111/2041-210X.12860

OECD [Organisation for Economic Co-operation and Development] (2010) Climate Change and Agriculture: Impacts, Adaptation and Mitigation. Online: go.nature.com/EFMuF5

OECD [Organisation for Economic Co-operation and Development] (2016) Farm Management Practices to Foster Green Growth, 160 p. Online: www.oecd-ilibrary.org/content/publication/9789264238657-en

Ogle, S. M., Breidt, F. J., Paustian, K. (2005) Agricultural management impacts on soil organic carbon storage under moist and dry climatic conditions of temperate and tropical regions. Biogeochemistry 72(1):87-121. doi: 10.1007/s10533-004-0360-2

Ogle, S. M., Buendia, L., Butterbach-Bahl, K., Breidt, F. J., Hartman, M., Yagi, K., Nayamuth, R., Spencer, S., Wirth, T., Smith, P. (2013) Advancing national greenhouse gas inventories for agriculture in developing countries: improving activity data, emission factors and software technology. Environmental Research Letters 8(1):15030. doi: 10.1088/1748-9326/8/1/015030

Ogle, S. M., Swan, A., Paustian, K. (2012) No-till management impacts on crop productivity, carbon input and soil carbon sequestration. Agriculture, Ecosystems & Environment 149:37-49. doi: 10.1016/j.agee.2011.12.010

Ohiri, A. C., Ezumah, H. C. (1990) Tillage effects on cassava (Manihot esculenta) production and some soil properties. Soil and Tillage Research 17(3-4):221-229. doi: 10.1016/0167-1987(90)90037-E

Olson, K. R., Al-Kaisi, M. M., Lal, R., Lowery, B. (2014) Examining the paired comparison method approach for determining soil organic carbon sequestration rates. Journal of Soil and Water Conservation 69(6):193A-197A. doi: 10.2489/jswc.69.6.193A

Omara, P., Aula, L., Oyebiyi, F., Raun, W. R. (2019) World Cereal Nitrogen Use Efficiency Trends: Review and Current Knowledge. Agrosystems, Geosciences & Environment 2(1):1-8. doi: 10.2134/age2018.10.0045

Osterburg, B., Heidecke, C., Bolte, A. (2019) Folgenabschätzung für Maßnahmenoptionen im Bereich Landwirtschaft und landwirtschaftliche Landnutzung, Forstwirtschaft und Holznutzung zur Umsetzung des Klimaschutzplans 2050. Johann Heinrich von Thünen-Institut

Osterburg, B., Rüter, S., Freibauer, A. (2013) Handlungsoptionen für den Klimaschutz in der deutschen Agrar- und Forstwirtschaft. Johann Heinrich von Thünen-Institut

Ostwald, M., Henders, S. (2014) Making two parallel land-use sector debates meet: Carbon leakage and indirect land-use change. Land Use Policy 36:533-542. doi: 10.1016/j.landusepol.2013.09.012

Palley, T. (2021) The economics of new developmentalism: a critical assessment. Investigación Económica 80(317):3. doi: 10.22201/fe.01851667p.2021.317.79804

Palmer, C. (2011) Property rights and liability for deforestation under REDD+: Implications for 'permanence' in policy design. Ecological Economics 70(4):571-576. doi: 10.1016/j.ecolecon.2010.10.011

Pan, W., Kim, M.-K., Ning, Z., Yang, H. (2020) Carbon leakage in energy/forest sectors and climate policy implications using meta-analysis. Forest Policy and Economics 115:102161. doi: 10.1016/j.forpol.2020.102161

Panison, F., Sangoi, L., Durli, M. M., Leolato, L. S., Coelho, A. E., Kuneski, H. F., Liz, V. O. de (2019) Timing and splitting of nitrogen side-dress fertilization of early corn hybrids for high grain yield. Revista Brasileira de Ciência do Solo 43. doi: 10.1590/18069657rbcs20170338

Parkin, T. B., Hatfield, J. L. (2010) Influence of nitrapyrin on N2O losses from soil receiving fall-applied anhydrous ammonia. Agriculture, Ecosystems & Environment 136(1-2):81-86. doi: 10.1016/j.agee.2009.11.014

Parkin, T. B., Hatfield, J. L. (2014) Enhanced efficiency fertilizers: effect on nitrous oxide emissions in Iowa. Agronomy Journal 106(2):694-702. doi: 10.2134/agronj2013.0219

Paustian, M., Theuvsen, L. (2017) Adoption of precision agriculture technologies by German crop farmers. Precision Agriculture 18(5):701-716. doi: 10.1007/s11119-016-9482-5

Pellerin, S., Bamière, L., Angers, D., Béline, F., Benoit, M., Butault, J.-P., Chenu, C., Colnenne-David, C., Cara, S. de, Delame, N., Doreau, M., Dupraz, P., Faverdin, P., Garcia-Launay, F., Hassouna, M., Hénault, C., Jeuffroy, M.-H., Klumpp, K., Metay, A., Moran, D., Recous, S., Samson, E., Savini, I., Pardon, L., Chemineau, P. (2017) Identifying cost-competitive greenhouse gas mitigation potential of French agriculture. Environmental Science & Policy 77:130-139. doi: 10.1016/j.envsci.2017.08.003

Perakis, S. S., Compton, J. E., Hedin, L. O. (2005) Nitrogen retention across a gradient of 15 N additions to an unpolluted temperate forest soil in Chile. Ecology 86(1):96-105. doi: 10.1890/04-0415

Pérez Domínguez, I., Fellmann, T., Witzke, P., Weiss, F., Hristov, J., Himics, M., Barreiro-Hurlé, J., Gómez-Barbero, M., Leip, A. (2020) Economic assessment of GHG mitigation policy options for EU agriculture: a closer look at mitigation options and regional mitigation costs - EcAMPA 3, Europäische Gemeinschaften, 89 p. JRC technical report

Petersen, S. O., Blanchard, M., Chadwick, D., del Prado, A., Edouard, N., Mosquera, J., Sommer, S. G. (2013) Manure management for greenhouse gas mitigation. Animal : an international journal of animal bioscience 7 Suppl 2:266-282. doi: 10.1017/S1751731113000736

Pike, T. (2008) Understanding Behaviours in a farming context: Bringing theoretical and applied evidence together from across Defra and highlighting policy relevance and implications for future research paper.: Defra Agricultural Change and Environment Observatory Discussion

Poeplau, C., Don, A. (2015) Carbon sequestration in agricultural soils via cultivation of cover crops – A meta-analysis. Agriculture, Ecosystems & Environment 200:33-41. doi: 10.1016/j.agee.2014.10.024

Povellato, A., Bosello, F., Giupponi, C. (2007) Cost-effectiveness of greenhouse gases mitigation measures in the European agro-forestry sector: a literature survey. Environmental Science & Policy 10(5):474-490. doi: 10.1016/j.envsci.2007.02.005

Powlson, D. S., Whitmore, A. P., Goulding, K. W. T. (2011) Soil carbon sequestration to mitigate climate change: a critical re-examination to identify the true and the false. European Journal of Soil Science 62(1):42-55. doi: 10.1111/j.1365-2389.2010.01342.x

Pugesgaard, S., Schelde, K., Larsen, S. U., Laerke, P. E., Jørgensen, U. (2015) Comparing annual and perennial crops for bioenergy production - influence on nitrate leaching and energy balance. GCB Bioenergy 7(5):1136-1149. doi: 10.1111/gcbb.12215

Pulleman, M. M., Bouma, J., van Essen, E. A., Meijles, E. W. (2000) Soil organic matter content as a function of different land use history. Soil Science Society of America Journal 64(2):689-693. doi: 10.2136/sssaj2000.642689x

Quiroga, A., Funaro, D., Noellemeyer, E., Peinemann, N. (2006) Barley yield response to soil organic matter and texture in the Pampas of Argentina. Soil and Tillage Research 90(1-2):63-68. doi: 10.1016/j.still.2005.08.019

Ragnauth, S. A., Creason, J., Alsalam, J., Ohrel, S., Petrusa, J. E., Beach, R. H. (2015) Global mitigation of non-CO 2 greenhouse gases: marginal abatement costs curves and abatement potential through 2030. Journal of Integrative Environmental Sciences 12(sup1):155-168. doi: 10.1080/1943815X.2015.1110182

Rainbow, R., Derpsch, R. (2011) Advances in No-Till Farming Technologies and Soil Compaction Management in Rainfed Farming Systems. Online: 10.1007/978-1-4020-9132-2_39

Reay, D. S., Davidson, E. A., Smith, K. A., Smith, P., Melillo, J. M., Dentener, F., Crutzen, P. J. (2012) Global agriculture and nitrous oxide emissions. Nature Climate Change 2(6):410-416. doi: 10.1038/nclimate1458

Rech, I., Polidoro, J. C., Pavinato, P. S. (2017) Additives incorporated into urea to reduce nitrogen losses after application to the soil. Pesquisa Agropecuária Brasileira 52(3):194-204. doi: 10.1590/s0100-204x2017000300007

Reicosky, D. (ed) (2018) Managing soil health for sustainable agriculture. Cambridge, UK: Burleigh Dodds Science Publishing Limited, 334 p. Burleigh Dodds series in agricultural science number 48

Reid, S., Smit, B., Caldwell, W., Belliveau, S. (2007) Vulnerability and adaptation to climate risks in Ontario agriculture. Mitigation and Adaptation Strategies for Global Change 12(4):609-637. doi: 10.1007/s11027-006-9051-8

Roberts, R. K., English, B. C., Larson, J. A., Cochran, R. L., Goodman, W. R., Larkin, S. L., Marra, M. C., Martin, S. W., Shurley, W. D., Reeves, J. M., et al. (2004) Adoption of Site-Specific Information and Variable-Rate Technologies in Cotton Precision Farming. Journal of Agricultural and Applied Economics 36(1):143-158. doi: 10.1017/S107407080002191X

Robertson, G., Grace, P. (2004) Greenhouse gas fluxes in tropical and temperate agriculture: the need for a full-cost accounting of Global Warming Potentials. In: Tropical Agriculture in Transition — Opportunities for Mitigating Greenhouse Gas Emissions? Springer, Dordrecht: pp 51-63. Online: link.springer.com/chapter/10.1007/978-94-017-3604-6_3

Robertson, G. P., Vitousek, P. M. (2009) Nitrogen in Agriculture: Balancing the Cost of an Essential Resource. Annual Review of Environment and Resources 34(1):97-125. doi: 10.1146/annurev.environ.032108.105046

Roger-Estrade, J., Richard, G., Dexter, A. R., Boizard, H., Tourdonnet, S., Bertrand, M., Caneill, J. (2009) Integration of soil structure variations with time and space into models for crop management. A review. Agronomy for Sustainable Development 29(1):135-142. doi: 10.1051/agro:2008052

Rørstad, P. K., Vatn, A., Kvakkestad, V. (2007) Why do transaction costs of agricultural policies vary? Agricultural Economics 36(1):1-11. doi: 10.1111/j.1574-0862.2007.00172.x

Rosenzweig, C., Tubiello, F. N. (2007) Adaptation and mitigation strategies in agriculture: an analysis of potential synergies. Mitigation and Adaptation Strategies for Global Change 12(5):855-873. doi: 10.1007/s11027-007-9103-8

Rotz, C. A., Taube, F., Russelle, M. P., Oenema, J., Sanderson, M. A., Wachendorf, M. (2005) Whole-Farm Perspectives of Nutrient Flows in Grassland Agriculture. Crop Science 45(6):2139-2159. doi: 10.2135/cropsci2004.0523

Roy, A. K., Wagner-Riddle, C., Deen, B., Lauzon, J., Bruulsema, T. (2014) Nitrogen application rate, timing and history effects on nitrous oxide emissions from corn (Zea mays L.). Canadian Journal of Soil Science 94(4):563-573. doi: 10.4141/cjss2013-118

Ruffo, M. L., Bollero, G. A. (2003) Modeling rye and hairy vetch residue decomposition as a function of degree-days and decomposition-days. Agronomy Journal 95(4):900-907. doi: 10.2134/agronj2003.9000

Samsonstuen, S., Åby, B. A., Crosson, P., Beauchemin, K. A., Aass, L. (2020) Mitigation of greenhouse gas emissions from beef cattle production systems. Acta Agriculturae Scandinavica, Section A — Animal Science 69(4):220-232. doi: 10.1080/09064702.2020.1806349

Sánchez, B., Iglesias, A., McVittie, A., Álvaro-Fuentes, J., Ingram, J., Mills, J., Lesschen, J. P., Kuikman, P. J. (2016) Management of agricultural soils for greenhouse gas mitigation: Learning from a case study in NE Spain. Journal of environmental management 170:37-49. doi: 10.1016/j.jenvman.2016.01.003

Sanz-Cobena, A., Lassaletta, L., Aguilera, E., Prado, A. d., Garnier, J., Billen, G., Iglesias, A., Sánchez, B., Guardia, G., Abalos, D., Plaza-Bonilla, D., Puigdueta-Bartolomé, I., Moral, R., Galán, E., Arriaga, H., Merino, P., Infante-Amate, J., Meijide, A., Pardo, G., Álvaro-Fuentes, J., Gilsanz, C., Báez, D., Doltra, J., González-Ubierna, S., Cayuela, M. L., Menéndez, S., Díaz-Pinés, E., Le-Noë, J., Quemada, M., Estellés, F., Calvet, S., van Grinsven, H., Westhoek, H., Sanz, M. J., Gimeno, B. S., Vallejo, A., Smith, P. (2017) Strategies for greenhouse gas emissions mitigation in Mediterranean agriculture: A review. Agriculture, Ecosystems & Environment 238:5-24. doi: 10.1016/j.agee.2016.09.038

Sawyer, J., Mallarino, A. (2008) Nutrient removal when harvesting corn stover, Iowa State University. Online: lib.dr.iastate.edu/cropnews/1006/

Sawyer, J., Nafziger, E., Randall, G., Bundy, L., Rehm, G., Joern, B. (2006) Concepts and Rationale for Regional Nitrogen Rate Guidelines for Corn. Online: publications.iowa.gov/id/eprint/3847

Sayre, R., Karagulle, D., Frye, C., Boucher, T., Wolff, N. H., Breyer, S., Wright, D., Martin, M., Butler, K., van Graafeiland, K., Touval, J., Sotomayor, L., McGowan, J., Game, E. T., Possingham, H. (2020) An assessment of the representation of ecosystems in global protected areas using new maps of World Climate Regions and World Ecosystems. Global Ecology and Conservation 21:e00860. doi: 10.1016/j.gecco.2019.e00860

Scharf, P. C., Wiebold, W. J., Lory, J. A. (2002) Corn yield response to nitrogen fertilizer timing and deficiency level. Agronomy Journal 94(3):435-441. doi: 10.2134/agronj2002.4350

Schimmelpfennig, D. (2016) Farm Profits and Adoption of Precision Agriculture

Schlesinger, W., Hartley, A. (1992) A global budget for atmospheric NH3. Biogeochemistry 15(3):191-211. doi: 10.1007/BF00002936

Schneider, U. A., McCarl, B. A. (2006) Appraising agricultural greenhouse gas mitigation potentials: effects of alternative assumptions. Agricultural Economics 35(3):277-287. doi: 10.1111/j.1574-0862.2006.00162.x

Schneider, U. A., McCarl, B. A., Schmid, E. (2007) Agricultural sector analysis on greenhouse gas mitigation in US agriculture and forestry. Agricultural Systems 94(2):128-140. doi: 10.1016/j.agsy.2006.08.001

Shepherd, T. G., Saggar, S., Newman, R. H., Ross, C. W., Dando, J. L. (2001) Tillage-induced changes to soil structure and organic carbon fractions in New Zealand soils. Soil Research 39(3):465. doi: 10.1071/SR00018

Shigueru Okumura, R., Takeo Yano, G., Cinque Mariano, D. de, Contandor Zaccheo, P., Wilson Takahashi, H. (2013) Nutrição nitrogenada no milho fertilizado com ureia tratada com inibidor de urease 34:157-169

Silva, A. G. B., Sequeira, C. H., Sermarini, R. A., Otto, R. (2017) Urease inhibitor NBPT on ammonia volatilization and crop productivity: a meta-analysis. Agronomy Journal 109(1):1-13. doi: 10.2134/agronj2016.04.0200

Siqueira Neto, M., Piccolo, M. d. C., Venzke Filho, S. d. P., Feigl, B. J., Cerri, C. C. (2010) Mineralização e desnitrificação do nitrogênio no solo sob sistema plantio direto. Bragantia 69(4):923-936. doi: 10.1590/S0006-87052010000400019

Smallbone, A., Jia, B., Atkins, P., Roskilly, A. P. (2020) The impact of disruptive powertrain technologies on energy consumption and carbon dioxide emissions from heavy-duty vehicles. Energy Conversion and Management: X 6:100030. doi: 10.1016/j.ecmx.2020.100030

Smil, V. (1999) Nitrogen in crop production: An account of global flows. Global Biogeochemical Cycles 13(2):647-662. doi: 10.1029/1999GB900015

Smith, P. (2016) Soil carbon sequestration and biochar as negative emission technologies. Global change biology 22(3):1315-1324. doi: 10.1111/gcb.13178

Smith, P., Haberl, H., Popp, A., Erb, K.-H., Lauk, C., Harper, R., Tubiello, F. N., Siqueira Pinto, A. de, Jafari, M., Sohi, S., Masera, O., Böttcher, H., Berndes, G., Bustamante, M., Ahammad, H., Clark, H., Dong, H., Elsiddig, E. A., Mbow, C., Ravindranath, N. H., Rice, C. W., Robledo Abad, C., Romanovskaya, A., Sperling, F., Herrero, M., House, J. I., Rose, S. (2013) How much land-based greenhouse gas mitigation can be achieved without compromising food security and environmental goals? Global change biology 19(8):2285-2302. doi: 10.1111/gcb.12160

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C. (2007) Policy and technological constraints to implementation of greenhouse gas mitigation options in agriculture. Agriculture, Ecosystems & Environment 118(1-4):6-28. doi: 10.1016/j.agee.2006.06.006

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M., Smith, J. (2008) Greenhouse gas mitigation in agriculture. Philosophical transactions of the Royal Society of London. Series B, Biological sciences 363(1492):789-813. doi: 10.1098/rstb.2007.2184

Snapp, S. S., Swinton, S. M., Labarta, R., Mutch, D., Black, J. R., Leep, R., Nyiraneza, J., O'Neil, K. (2005) Evaluating Cover Crops for Benefits, Costs and Performance within Cropping System Niches. Agronomy Journal 97(1):322-332. doi: 10.2134/agronj2005.0322a

Snyder, C. S., Bruulsema, T. W., Jensen, T. L., Fixen, P. E. (2009) Review of greenhouse gas emissions from crop production systems and fertilizer management effects. Agriculture, Ecosystems & Environment 133(3-4):247-266. doi: 10.1016/j.agee.2009.04.021

Soane, B.D (ed) (1994) Soil compaction in crop production. Amsterdam: Elsevier, 662 p. Developments in Agricultural Engineering 11

Sørensen, L. H. (1987) Organic matter and microbial biomass in a soil incubated in the field for 20 years with 14C-labelled barley straw. Soil Biology and Biochemistry 19(1):39-42. doi: 10.1016/0038-0717(87)90123-4

Soumare, A., Diedhiou, A. G., Thuita, M., Hafidi, M., Ouhdouch, Y., Gopalakrishnan, S., Kouisni, L. (2020) Exploiting biological nitrogen fixation: a route towards a sustainable agriculture. Plants (Basel, Switzerland) 9(8). doi: 10.3390/plants9081011

Spangler, K. R., Lynch, A. H., Spera, S. A. (2017) Precipitation Drivers of Cropping Frequency in the Brazilian Cerrado: Evidence and Implications for Decision-Making. Weather, Climate, and Society 9(2):201-213. doi: 10.1175/WCAS-D-16-0024.1

Stein, L. Y., Klotz, M. G. (2016) The nitrogen cycle. Current biology : CB 26(3):R94-8. doi: 10.1016/j.cub.2015.12.021

Stevens, C. J., Quinton, J. N. (2009) Policy implications of pollution swapping. Physics and Chemistry of the Earth, Parts A/B/C 34(8-9):589-594. doi: 10.1016/j.pce.2008.01.001

Stivers-Young, L. J., Tucker, F. A. (1999) Cover-cropping practices of vegetable producers in western New York. HortTechnology 9(3):459-465. doi: 10.21273/HORTTECH.9.3.459

Stockmann, U., Adams, M. A., Crawford, J. W., Field, D. J., Henakaarchchi, N., Jenkins, M., Minasny, B., McBratney, A. B., Courcelles, V. R. de, Singh, K., Wheeler, I., Abbott, L., Angers, D. A., Baldock, J., Bird, M., Brookes, P. C., Chenu, C., Jastrow, J. D., Lal, R., Lehmann, J., O'Donnell, A. G., Parton, W. J., Whitehead, D., Zimmermann, M. (2013) The knowns, known unknowns and unknowns of sequestration of soil organic carbon. Agriculture, Ecosystems & Environment 164:80-99. doi: 10.1016/j.agee.2012.10.001

Stott, D. E., Elliott, L. F., Papendick, R. I., Campbell, G. S. (1986) Low temperature or low water potential effects on the microbial decomposition of wheat residue. Soil Biology and Biochemistry 18(6):577-582. doi: 10.1016/0038-0717(86)90078-7

Stuart, D., Schewe, R. L. (2016) Constrained choice and climate change mitigation in US agriculture: structural barriers to a climate change ethic. Journal of Agricultural and Environmental Ethics 29(3):369-385. doi: 10.1007/s10806-016-9605-z

Sutton, M. A., Oenema, O., Erisman, J. W., Leip, A., van Grinsven, H., Winiwarter, W. (2011) Too much of a good thing. Nature 472(7342):159-161. doi: 10.1038/472159a

Swinton, S. M., Rector, N., Philip Robertson, G., Jelejole-Foreman, C. B., Lupi, F. (2015) Farmer decisions about adopting environmentally beneficial practices.: n The Ecology of Agricultural Landscapes: Long-Term Research on the Path to Sustainability:340-359

Tang, K., Ma, C. (2022) The cost-effectiveness of agricultural greenhouse gas reduction under diverse carbon policies in China. China Agricultural Economic Review 14(4):758-773. doi: 10.1108/CAER-01-2022-0008

Tenuta, M., Gao, X., Flaten, D. N., Amiro, B. D. (2016) Lower nitrous oxide emissions from anhydrous ammonia application prior to soil freezing in late fall than spring pre-plant application. Journal of Environmental Quality 45(4):1133-1143. doi: 10.2134/jeq2015.03.0159

Teshager, A. D., Gassman, P. W., Secchi, S., Schoof, J. T. (2017) Simulation of targeted pollutant-mitigation-strategies to reduce nitrate and sediment hotspots in agricultural watershed. The Science of the total environment 607-608:1188-1200. doi: 10.1016/j.scitotenv.2017.07.048

Thapa, R., Chatterjee, A., Awale, R., McGranahan, D. A., Daigh, A. (2016) Effect of Enhanced Efficiency Fertilizers on Nitrous Oxide Emissions and Crop Yields: A Meta-analysis. Soil Science Society of America Journal 80(5):1121-1134. doi: 10.2136/sssaj2016.06.0179

Thapa, R., Poffenbarger, H., Tully, K. L., Ackroyd, V. J., Kramer, M., Mirsky, S. B. (2018) Biomass production and nitrogen accumulation by hairy vetch-cereal rye mixtures: a meta-analysis. Agronomy Journal 110(4):1197-1208. doi: 10.2134/agronj2017.09.0544

Tiecher, T., Gubiani, E., Santanna, M. A., Veloso, M. G., Calegari, A., Canalli, L. B. d. S., Finckh, M. R., Caner, L., Rheinheimer, D. D. S. (2020) Effect of 26-years of soil tillage systems and winter cover crops on C and N stocks in a Southern Brazilian Oxisol. Revista Brasileira de Ciência do Solo 44. doi: 10.36783/18069657rbcs20200029

Timilsena, Y. P., Adhikari, R., Casey, P., Muster, T., Gill, H., Adhikari, B. (2015) Enhanced efficiency fertilisers: a review of formulation and nutrient release patterns. Journal of the Science of Food and Agriculture 95(6):1131-1142. doi: 10.1002/jsfa.6812

Timma, L., Dace, E., Trydeman Knudsen, M. (2020) Temporal aspects in emission accounting—case study of agriculture sector. Energies 13(4):800. doi: 10.3390/en13040800

Tonitto, C., David, M. B., Drinkwater, L. E. (2006) Replacing bare fallows with cover crops in fertilizer-intensive cropping systems: A meta-analysis of crop yield and N dynamics. Agriculture, Ecosystems & Environment 112(1):58-72. doi: 10.1016/j.agee.2005.07.003

Tubiello, F. N., Rahman, A., Mann, W., Schmidhuber, J., Koleva, M., Müller, A. (2009) Carbon financial mechanisms for agriculture and rural development: challenges and opportunities along the Bali roadmap. Climatic Change 97(1-2):3-21. doi: 10.1007/s10584-009-9611-5

Tully, K., Ryals, R. (2017) Nutrient cycling in agroecosystems: Balancing food and environmental objectives. Agroecology and Sustainable Food Systems 41(7):761-798. doi: 10.1080/21683565.2017.1336149

U.S. Energy Information Administration (2020). Online: www.eia.gov/state/seds/seds-data-complete.php?sid=US#Production. Accessed: 09.02.2021

UNEP [United Nations Environment Programme] (1998) Economics of greenhouse gas limitations. Main reports: methodological guidelines

USDA [United States Department of Agriculture] (2020) National Agricultural Statistics Service. Online: www.nass.usda.gov/Publications/AgCensus/2017/Online_Resources/County_Profiles/Iowa/index.php. Accessed: 16.06.2020

USDA [United States Department of Agriculture] (2021a) Economic Research Service: State Export Data. Online: www.ers.usda.gov/data-products/state-agricultural-trade-data/state-agricultural-trade-data/. Accessed: 09.02.2021

USDA [United States Department of Agriculture] (2021b) Economic Research Service: U.S. Bioenergy Statistics. Online: www.ers.usda.gov/data-products/us-bioenergy-statistics/. Accessed: 09.02.2021

van Spanning, R. J. M., M. J. Delgado, D. J. Richardson (2005) The Nitrogen Cycle: Denitrification and its Relationship to N2 Fixation. In: Nitrogen Fixation in Agriculture, Forestry, Ecology, and the Environment. Springer, Dordrecht: pp 277-342. Online: link.springer.com/chapter/10.1007/1-4020-3544-6_13

van Vuuren, D. P., Hoogwijk, M., Barker, T., Riahi, K., Boeters, S., Chateau, J., Scrieciu, S., van Vliet, J., Masui, T., Blok, K., Blomen, E., Kram, T. (2009) Comparison of top-down and bottom-up estimates of sectoral and regional greenhouse gas emission reduction potentials. Energy Policy 37(12):5125-5139. doi: 10.1016/j.enpol.2009.07.024

Venterea, R. T., Rolston, D. E., Cardon, Z. G. (2005) Effects of soil moisture, physical, and chemical characteristics on abiotic nitric oxide production. Nutrient Cycling in Agroecosystems 72(1):27-40. doi: 10.1007/s10705-004-7351-5

Vermont, B., Cara, S. de (2010) How costly is mitigation of non-CO2 greenhouse gas emissions from agriculture? Ecological Economics 69(7):1373-1386. doi: 10.1016/j.ecolecon.2010.02.020

Verspecht, A., Vandermeulen, V., Avest, E. ter, van Huylenbroeck, G. (2012) Review of trade-offs and co-benefits from greenhouse gas mitigation measures in agricultural production. Journal of Integrative Environmental Sciences 9(sup1):147-157. doi: 10.1080/1943815X.2012.698989

Vetsch, J. A., Randall, G. W., Lamb, J. A. (2007) Corn and Soybean Production as Affected by Tillage Systems. Agronomy Journal 99(4):952-959. doi: 10.2134/agronj2006.0149

Viero, F., Bayer, C., Fontoura, S. M. V., Moraes, R. P. de (2014) Ammonia volatilization from nitrogen fertilizers in no-till wheat and maize in southern Brazil. Revista Brasileira de Ciência do Solo 38(5):1515-1525. doi: 10.1590/S0100-06832014000500017

Viero, F., Menegati, G. B., Carniel, E., Da Silva, P. R. F., Bayer, C. (2017) Urease inhibitor and irrigation management to mitigate ammonia volatilization from urea in no-till corn. Revista Brasileira de Ciência do Solo 41(0). doi: 10.1590/18069657rbcs20160567

Vollmer, E., Mußhoff, O. (2018) Average protein content and its variability in winter wheat: a forecast model based on weather parameters. Earth Interactions 22(19):1-24. doi: 10.1175/EI-D-18-0011.1

Walling, E., Vaneeckhaute, C. (2020) Greenhouse gas emissions from inorganic and organic fertilizer production and use: A review of emission factors and their variability. Journal of environmental management 276:111211. doi: 10.1016/j.jenvman.2020.111211

Walther, S. (2014) Determinants of competitiveness of agriholdings and independent farms in Ukrainian arable production. Braunschweig: Johann Heinrich von Thünen-Institut, 34 p. Thünen Report 15. Online: d-nb.info/1050725816/

Wang, R., Gamon, J. A. (2019) Remote sensing of terrestrial plant biodiversity. Remote Sensing of Environment 231:111218. doi: 10.1016/j.rse.2019.111218

Wayman, S., Cogger, C., Benedict, C., Burke, I., Collins, D., Bary, A. (2015) The influence of cover crop variety, termination timing and termination method on mulch, weed cover and soil nitrate in reduced-tillage organic systems. Renewable Agriculture and Food Systems 30(5):450-460. doi: 10.1017/S1742170514000246

West, T. O., Post, W. M. (2002) Soil organic carbon sequestration rates by tillage and crop rotation. Soil Science Society of America Journal 66(6):1930-1946. doi: 10.2136/sssaj2002.1930

Westra, P., Wilson, R. G., Miller, S. D., Stahlman, P. W., Wicks, G. W., Chapman, P. L., Withrow, J., Legg, D., Alford, C., Gaines, T. A. (2008) Weed population dynamics after six years under glyphosate- and conventional herbicide-based weed control strategies. Crop Science 48(3):1170. doi: 10.2135/cropsci2007.08.0490

Wicke, B., Verweij, P., van Meijl, H., van Vuuren, D. P., Faaij, A. P. C. (2012) Indirect land use change: review of existing models and strategies for mitigation. Biofuels 3(1):87-100. doi: 10.4155/bfs.11.154

Wójtowicz, M., Wójtowicz, A., Piekarczyk, J., others (2016) Application of remote sensing methods in agriculture. Communications in Biometry and Crop Science 11(1):31-50

Woli, K. P., Fernández, F. G., Sawyer, J. E., Stamper, J. D., Mengel, D. B., Barker, D. W., Hanna, M. H. (2014) Agronomic comparison of anhydrous ammonia applied with a high speed-low draft opener and conventional knife injection in corn. Agronomy Journal 106(3):881-892. doi: 10.2134/agronj13.0441

Wolt, J. D. (2000) Nitrapyrin behavior in soils and environmental considerations. Journal of Environmental Quality 29(2):367-379. doi: 10.2134/jeq2000.00472425002900020002x

Wolt, J. D. (2004) A meta-evaluation of nitrapyrin agronomic and environmental effectiveness with emphasis on corn production in the Midwestern USA. Nutrient Cycling in Agroecosystems 69(1):23-41. doi: 10.1023/b:fres.0000025287.52565.99

World Bank (2021) Global Economic Monitor (GEM). Online: datacatalog.worldbank.org/search/dataset/0037798. Accessed: 08.09.2021

World Bank (2022) State and Trends of Carbon Pricing 2022. Online: openknowledge.worldbank.org/handle/10986/37455?show=full

Wreford, A., Ignaciuk, A., Gruère, G. (2017) Overcoming barriers to the adoption of climate-friendly practices in agriculture, hg. v. Organisation for Economic Co-operation and Development (OECD), 41 p. OECD Food, Agriculture and Fisheries Papers

WRI [World Resources Institute] (2022) Climate Analysis Indicators Tool (CAIT) - Climate Data Explorer. Online: www.climatewatchdata.org. Accessed: 06.04.2022

Xu, J., Gao, J., Holanda, H. V. de, Rodríguez, L. F., Caixeta-Filho, J. V., Zhong, R., Jiang, H., Li, H., Du, Z., Wang, X., Wang, S., Ting, K. C., Ying, Y., Lin, T. (2021) Double cropping and cropland expansion boost grain production in Brazil. Nature Food 2(4):264-273. doi: 10.1038/s43016-021-00255-3

Yadav, S. N. (1997) Formulation and estimation of nitrate-nitrogen leaching from corn cultivation. Journal of Environmental Quality 26(3):808-814. doi: 10.2134/jeq1997.00472425002600030031x

Yadav, S. N., Peterson, W., Easter, K. W. (1997) Do farmers overuse nitrogen fertilizer to the detriment of the environment? Environmental & Resource Economics 9(3):323-340. doi: 10.1007/BF02441403

Yan, H., Xie, L., Guo, L., Fan, J., Diao, T., Lin, M., Zhang, H., Lin, E. (2014) Characteristics of nitrous oxide emissions and the affecting factors from vegetable fields on the North China Plain. Journal of environmental management 144:316-321. doi: 10.1016/j.jenvman.2014.06.004

Yin, R. K. (1994) Discovering the future of the case study method in evaluation research. Evaluation Practice 15(3):283-290. doi: 10.1016/0886-1633(94)90023-x

Zhang, W.-F., Dou, Z.-X., He, P., Ju, X.-T., Powlson, D., Chadwick, D., Norse, D., Lu, Y.-L., Zhang, Y., Wu, L., Chen, X.-P., Cassman, K. G., Zhang, F.-S. (2013) New technologies reduce greenhouse gas emissions from nitrogenous fertilizer in China. Proceedings of the National Academy of Sciences 110(21):8375-8380. doi: 10.1073/pnas.1210447110

Zhang, X., Davidson, E. A., Mauzerall, D. L., Searchinger, T. D., Dumas, P., Shen, Y. (2015a) Managing nitrogen for sustainable development. Nature 528(7580):51-59. doi: 10.1038/nature15743

Zhang, X., Mauzerall, D. L., Davidson, E. A., Kanter, D. R., Cai, R. (2015b) The economic and environmental consequences of implementing nitrogen-efficient technologies and management practices in agriculture. Journal of Environmental Quality 44(2):312-324. doi: 10.2134/jeq2014.03.0129

Zhiming Qi, Matthew Justin Helmers, Peter A Lawlor (2008) Effect of different land covers on nitrate-nitrogen leaching and nitrogen uptake in Iowa. In: American Society of Agricultural and Biological Engineers: p 1

Zimmer, Y., Deblitz, C. (2005) agri benchmark Cash Crop: A standard operating procedure to define typical farms, agri benchmark, 19 p. Online: www.agribenchmark.org/fileadmin/Dateiablage/B-Cash-Crop/Projects/SOP-cashcrop-0512.pdf

List of appendices

Appendix A: values and assumptions for the calculation of GHG emissions and costs

List of tables

Table A.1 Input costs in baseline scenario in Typical Farms (USD)

Input	Unit	Iowa (US)	Paraná (BR)	Mecklenburg (DE)
Diesel	liter	0.61	0.96	0.97
Labor	hour	15.00	6.59	22.74
Anhydrous ammonia (82% N)	kg prod.	0.64	-	-
Urea (46%)	kg prod.	-	0.38	0.32
Triple superphosphate (46% P_2O_5)	kg prod.	0.37	-	-
Potassium chloride (60% K_2O)	kg prod.	0.38	0.46	0.37
Monoammonium phosphate (11% N - 52% P_2O_5)	kg prod.	-	0.58	-
Diammonium phosphate (18% N - 28% P_2O_5)	kg prod.	-	-	0.45
Urea ammoniumnitrate (28% N)	kg prod.	-	-	0.21
Dolomite	t prod.	-	33.98	-
Seed	ha	245.37 (corn) 127.82 (soybean)	261.35	60.27
Crop care	ha	75.70 (corn) 69.19 (soybean)	162.62	227.44
Machinery	ha	356.90 (corn) 266.50 (soybean)	-	167.01

Source: own calculation based on focus groups.

Table A.2 Summary of emission factors (EF) and coefficients for GHG calculation

Coefficient/Emission factor	Unit	Source	Iowa (US)	Paraná (BR)	Mecklenburg (DE)
EF_{diesel}: EF for diesel	kg CO_{2eq}/l	(1)	3.13	3.13	3.13
EF_{SN}: EF for N_2O from synthetic N input	kg N_2O-N/kg N	(2)	0.016	0.016	0.016
EF_{CR}: EF for N_2O from N from crop residues	kg N_2O-N/kg N	(2)	0.006	0.006	0.006
Yield	t/ha	(2)	12.67 (corn) 3.81 (soybean)	13.00	9.00
Moisture: moisture content of the grain	%	(3)	15.5 (corn) 13 (soybean)	14	14
Slope	-	(2)	1.03 (corn) 0.93 (soybean)	1.03	1.61
Intercept		(2)	0.61 (corn) 1.35 (soybean)	0.61	0.4
NC_{AG}: nitrogen content in above-ground residues	kg N/kg DM	(2)	0.006 (corn) 0.008 (soybean)	0.006	0.006
RS: ratio of below-ground to above-ground biomass	-	(2)	0.22 (corn) 0.19 (soybean)	0.22	0.23
NC_{BG}: nitrogen content in below-ground residues	kg N/kg DM	(2)	0.007 (corn) 0.008 (soybean)	0.007	0.009
$Frac_{GASF}$: fraction of nitrogen volatilized as NH_3 and NO_X	kg N vol./kg N	(2)	0.08 (a. ammonia)	0.15 (urea) 0.08 (MAP)	0.15 (urea) 0.1 (UAN) 0.08 (DAP)
$Frac_{LEACHG}$: fraction of nitrogen leached	kg N leach/kg N	(2)	0.24	0.24	0.24
Nitrogen content in harvested parts	kg N/kg DM	(4)	0.0131 (corn) 0.0657 (soybean)	0.0157	0.0219

Note: Multiply the value in N_2O-N by 44/28 to transform to N_2O. Multiply the value in C by 44/12 to transform to CO_2.
Sources: (1) various, see Chapter 3.5.5; (2) IPCC (2019a); (3) focus groups; (4) Sawyer and Mallarino (2008) for corn in Iowa, Ciampitti and Salvagiotti (2018) for soybeans in Iowa, Garcia et al.(2018a) for corn in Paraná; BMEL (2021a) for winter wheat in Mecklenburg-Vorpommern; (5) Brentrup et al. (2018).

Table A.2 Summary of emission factors (EF) and coefficients for GHG calculation (continued)

Coefficient/Emission factor	Unit	Source	Iowa (US)	Paraná (BR)	Mecklenburg (DE)
EF_{ATD}: EF for N2O from nitrogen deposited	kg N_2O-N/kg N	(2)	0.014	0.014	0.014
EF_L: EF for N2O from nitrogen leached	kg N_2O-N/kg N	(2)	0.011	0.011	0.011
$EFl_{limestone}$: EF limestone applied	kg C/t limestone	(2)	-	0.12	-
EF_{urea}: EF urea applied	kg C/kg urea	(2)	-	0.2	0.2
EF manufacture of limestone	kg CO_{2eq}/kg prod.	(5)	-	0.007	-
EF manufacture of anhydrous ammonia	kg CO_{2eq}/kg prod.	(5)	2.49	-	-
EF manufacture of urea	kg CO_{2eq}/kg prod.	(5)	1.01	1.01	0.88
EF manufacture monoammonium phosphate (MAP)	kg CO_{2eq}/kg prod.	(5)	-	0.52	-
EF manufacture diammonium phosphate (DAP)	kg CO_{2eq}/kg prod.	(5)	-	-	0.63
EF manufacture potassium chloride	kg CO_{2eq}/kg prod.	(5)	0.25	0.25	0.25
EF manufacture triple superphosphate	kg CO_{2eq}/kg prod.	(5)	0.18	-	-
EF manufacture urea ammonium nitrate (UAN)	kg CO_{2eq}/kg prod.	(5)	-	-	0.78

Note: Multiply the value in N_2O-N by 44/28 to transform to N_2O. Multiply the value in C by 44/12 to transform to CO_2.
Sources: (1) various, see Chapter 3.5.5; (2) IPCC (2019a); (3) focus groups; (4) Sawyer and Mallarino (2008) for corn in Iowa, Ciampitti and Salvagiotti (2018) for soybeans in Iowa, Garcia et al.(2018a) for corn in Paraná; BMEL (2021a) for winter wheat in Mecklenburg-Vorpommern; (5) Brentrup et al. (2018).

Table A.3 Considerations on evaluation of feasibility of mitigation strategies in Typical Farms

Mitigation strategy	Iowa (US)	Paraná (BR)	Mecklenburg-Vorpommern (DE)
Optimization of nitrogen fertilization rate	Evaluated	Evaluated	Evaluated
Utilization of organic fertilizer	Not evaluated: already used as fertilizer in the region, although not in the Typical Farm	Not evaluated: livestock production is not significant in the region	Not evaluated: already used as fertilizer in the region, although not in the Typical Farm
Use of legumes as cover crops	Evaluated	Evaluated	Not evaluated: no suitable cover crop was determined by the focus group
Enhanced-efficiency fertilizer	Evaluated	Evaluated	Not evaluated: urease inhibitor is part of baseline, but there is a lack of scientific data on the effect of combining more additives
Reduction of tillage intensity	Evaluated	Not evaluated: already part of the baseline scenario	Not evaluated: a literature review indicates a yield reduction if no-till is implemented*, risk of indirect land use change could not be properly calculated
Stopping crop residue burning	Not evaluated: already part of the baseline scenario	Not evaluated: already part of the baseline scenario	Not evaluated: already part of the baseline scenario
Cover crops (carbon sequestration)	Evaluated	Not evaluated: already part of the baseline scenario	Evaluated

*Based on the results published by Gruber et al. (2012) and Huynh et al. (2019)
Sources: own estimation.

Table A.4 Coefficients and emission factors used for cereal rye and hairy vetch in Typical Farm in Iowa

Coefficient/Emission factor	Unit	Source	Cereal rye	Hairy vetch
EF_{SN}: EF for N_2O from synthetic N input	kg N_2O-N/kg N	(1)	0.016	0.016
EF_{CR}: EF for N_2O from N from crop residues	kg N_2O-N/kg N	(1)	0.006	0.006
DRY: Moisture content	%	(1)	12	10
Slope	-	(1)	1.09	1.13
Intercept	-	(1)	0.88	0.85
$N_{AG(T)}$: nitrogen content in above-ground residues	kg N/kg DM	(1)	0.005	0.027
$RS_{(T)}$: ratio of below-ground biomass to above-ground biomass	-	(1)	0.22	0.4
$N_{BG(T)}$: nitrogen content in below-ground residues	kg N/kg DM	(1)	0.011	0.022
$Frac_{GASF}$: fraction of synthetic nitrogen volatilized as NH_3 and NO_X	kg N volatilized/kg N	(1)	0.08	0.08
$Frac_{LEACHG}$: fraction of nitrogen leached	kg N leached/kg N	(1)	0.24	0.24
EF_{ATD}: EF for N_2O from nitrogen deposited	kg N_2O-N/kg N	(1)	0.014	0.014
EF_L: EF for N_2O from nitrogen leached	kg N_2O-N/kg N	(1)	0.011	0.011
EF_{urea}: EF urea applied	kg C/kg urea	(1)	0.2	-
EF for manufacture of urea	kg CO_{2eq}/kg product	(2)	1.01	-
EF for manufacture of potassium chloride	kg CO_{2eq}/kg product	(2)	0.25	0.25
EF for manufacture of triple superphosphate	kg CO_{2eq}/kg product	(2)	0.18	0.18

Note: Multiply the value in N_2O-N by 44/28 to transform to N_2O. Multiply the value in C by 44/12 to transform to CO_2.
Sources: (1) IPCC (2019a); (2) Brentrup et al. (2018).

Table A.5 Coefficients used for nitrogen additions from cover crops in Typical Farm in Paraná and Mecklenburg-Vorpommern

Coefficient/Emission factor	Unit	Black oats (Paraná)	Common vetch (Paraná)	Oats (Mecklenburg)
Above-ground biomass	kg DM/ha	2,940	1,960	1,500
$N_{AG(T)}$: nitrogen content in above-ground residues	kg N/kg DM	0.007	0.007	0.007
$RS_{(T)}$: ratio of below-ground biomass to above-ground biomass	-	0.250	0.400	0.250
$N_{BG(T)}$: nitrogen content in below-ground residues	kg N/kg DM	0.008	0.022	0.008

Source: IPCC (2019a).

Table A.6 Coefficients and emission factors used for prevented leaching in corn silage in Typical Farm in Mecklenburg-Vorpommern

Coefficient/Emission factor	Unit	Source	Corn silage
EF_{SN}: EF for N_2O from synthetic N input	kg N_2O-N/kg N	(1)	0.016
$Frac_{GASF}$: fraction of synthetic nitrogen volatilized as NH_3 and NO_X	kg N volatilized/kg N	(1)	0.15
$Frac_{LEACHG}$: fraction of nitrogen leached	kg N leached/kg N	(1)	0.24
EF_{ATD}: EF for N_2O from nitrogen deposited	kg N_2O-N/kg N	(1)	0.014
EF_L: EF for N_2O from nitrogen leached	kg N_2O-N/kg N	(1)	0.011
EF_{urea}: EF urea applied	kg C/kg urea	(1)	0.2
EF for manufacture of urea	kg CO_{2eq}/kg product	(2)	1.01

Note: Multiply the value in N_2O-N by 44/28 to transform to N_2O. Multiply the value in C by 44/12 to transform to CO_2.
Sources: (1) IPCC (2019a), (2) Brentrup et al. (2018).

Table A.7 Input use in mitigation strategies in corn in Typical Farm in Iowa

Strategy	Anhydrous ammonia (82% N) (kg prod./ha)		Triple superphosphate (46% P_2O_5) (kg prod./ha)	Potassium chloride (60% K_2O) (kg prod./ha)	Diesel (l/ha)	Labor (h/ha)
Baseline (US-0)	220.42		173.04	105.67	78.22	2.47
Optimization of nitrogen fertilization (US-1)	201.70 190.51 214.32	(standard) (best) (worst)	173.04	105.67	83.83	2.60
Reduce tillage intensity (US-2)	220.42	(all cases)	173.04	105.67	58.58	1.92
Use of cover crops (US-3)	179.45 172.62 186.28	(standard) (best) (worst)	173.04	105.67	79.15	2.55
Use of nitrification inhibitors (US-4)	208.96 208.72 209.09	(standard) (best) (worst)	173.04	105.67	78.22	2.47
Combination of strategies (US-C)	153.35 136.89 169.82	(standard) (best) (worst)	173.04	105.67	65.12	2.14

Note: When no difference between the cases is shown, the same value is assumed in the three cases. The volume of diesel shown in this table portrays only the requirement by the operations conducted using the machinery of the Typical Farm. Hence, the diesel utilized by the contractor is not included in this value.
Source: own calculation based on focus groups.

Table A.8 Input use in mitigation strategies in soybean in Typical Farm in Iowa

Strategy	Triple superphosphate (46% P_2O_5) (kg prod./ha)	Potassium chloride (60% K_2O) (kg prod./ha)	Diesel (l/ha)	Labor (h/ha)
Baseline (US-0)	108.00	159.00	35.54	1.34
Reduce tillage intensity (US-2)	108.00	159.00	28.99	1.21
Use of cover crops (US-3)	108.00	159.00	36.47	1.43
Combination of strategies (US-C)	108.00	159.00	29.92	1.29

Note: When no differentiation between the cases is shown, the same value is assumed in the three cases. The volume of diesel shown in this table portrays only the requirement by the operations conducted using the machinery of the Typical Farm. Hence, the diesel utilized by the contractor is not included in this value.
Source: own calculation based on focus groups.

Table A.9 Input use in mitigation strategies in corn in Typical Farm in Paraná

Strategy	Urea (46% N) (kg prod./ha)		Monoammonium phosphate (11% N - 52% P_2O_5) (kg prod./ha)	Potassium chloride (60% K_2O) (kg prod./ha)	Limestone (t/ha)	Diesel (l/ha)	Labor (h/ha)
Baseline (BR-0)	410.00		290.00	150.00	1.00	20.00	1.50
Optimization of nitrogen fertilization (BR-1)	376.30	(standard)	290.00	150.00	1.00	20.00	1.50
	351.96	(best)					
	404.35	(worst)					
Use of legumes as cover crops (BR-2)	333.91	(standard)	290.00	150.00	1.00	20.80	1.60
	323.04	(best)					
	344.78	(worst)					
Use of urease inhibitors (BR-3)	399.78	(standard)	290.00	150.00	1.00	20.00	1.50
	397.61	(best)					
	400.87	(worst)					
Combination of strategies (BR-C)	295.20	(standard)	290.00	150.00	1.00	20.80	1.60
	261.10	(best)					
	332.10	(worst)					

Note: When no differentiation between the cases is shown, the same value is assumed in the three cases. The volume of diesel shown in this table portrays only the requirement by the operations conducted using the machinery of the Typical Farm. Hence, the diesel utilized by the contractor is not included in this value.
Source: own calculation based on focus groups.

Table A.10 **Input use in mitigation strategies in winter wheat in Typical Farm in Mecklenburg-Vorpommern**

Strategy	**Urea (46% N)**	**Diammonium phosphate (18% N - 28% P_2O_5)**	**Potassium chloride (60% K_2O)**	**Urea ammonium nitrate (28% N)**		**Diesel**	**Labor**
	(kg prod./ha)	**(kg prod./ha)**	**(kg prod./ha)**	**(kg prod./ha)**		**(l/ha)**	**(h/ha)**
Baseline (DE-0)	260.87	100.00	100.00	196.43		90.00	3.50
Optimization of nitrogen fertilization (DE-1)	260.87 260.87	100.00	100.00	184.25 144.64	(standard) (best)	90.00	3.50
Use of cover crops (DE-2)	260.87	100.00	100.00	196.43	(all cases)	96.50	3.64
Combination of strategies (DE-C)	260.87 260.87	100.00	100.00	184.25 144.64	(standard) (best)	96.50	3.64

Note: When no differentiation between the cases is shown, the same value is assumed in the three cases. The volume of diesel shown in this table portrays only the requirement by the operations conducted using the machinery of the Typical Farm. Hence, the diesel utilized by the contractor is not included in this value.
Source: own calculation based on focus groups.

Table A.11 Assumptions for the calculation of machinery costs in Typical Farm in Iowa

Type	Equity fixed assets (%)	Long term loan (%)	Long term deposit (%)	Purchase cost (USD)	Salvage value (USD)	Depr. period (years)	Repurchase cost (USD)	Repairs (USD/year)	Usage (ha/a or h/a)
Anhydrous ammonia applicator	85	5.5	2.0	45,000	15,000	10	70,000	950	364
Tractor	85	5.5	2.0	175,000	80,000	15	225,000	2,000	500
Corn stalk stompers	85	5.5	2.0	3,500	0	5	4,000	20	728
Disk ripper	85	5.5	2.0	45,000	22,500	8	50,000	750	364
Soil finisher	85	5.5	2.0	50,000	15,000	8	57,000	800	728
Pesticide sprayer	85	5.5	2.0	26,000	5,000	15	55,000	1,500	1,092

Source: Focus groups.

Table A.12 Assumptions for the calculation of machinery costs in Typical Farm in Mecklenburg-Vorpommern

Type	Equity fixed assets (%)	Long term loan (%)	Long term deposit (%)	Purchase cost (USD)	Salvage value (USD)	Depr. period (years)	Repurchase (USD)	Repairs (USD/year)	Usage (ha/a or h/a)
Seed drill (cultivator)	85	1.8	0.9	8,529	2,132	10	9,808	227	275
Tractor	85	1.8	0.9	102,349	45,488	8	125,094	1,365	800
Disk harrow	85	1.8	0.9	46,626	17,058	10	56,861	2,843	2,000

Source: Focus groups.

Table A.13 Change in labor, diesel, and machinery costs from farm operations in Typical Farms

Farm and strategy	Operation or change	Labor (h/ha)	Diesel consumption (l/ha)	Equipment affected	Depreciation (USD/ha)	Repairs (USD/ha)
Iowa Optimization of nitrogen fertilization (US-1)	Additional pass to spread nitrogen	0.13	5.6	Anhydrous ammonia applicator	-	2.62
				Tractor	-	0.52
Iowa Reduce tillage intensity (US-2)	Stopping disk ripper (corn)	0.42	13.09	Disk ripper	11.93	1.92
				Tractor	-	1.67
Iowa Reduce tillage intensity (US-2)	Stopping soil finisher (corn)	0.14	6.55	Soil finisher	8.34	1.1
				Tractor	-	0.55
Iowa Reduce tillage intensity (US-2)	Protection of wheels	-	-	Stalk stompers	1.19	0.03
Iowa Reduce tillage intensity (US-2)	Stopping soil finisher (soybean)	0.14	6.55	Soil finisher	8.34	1.1
				Tractor		0.55
Iowa Use of cover crops (US-3)	Termination of cover crop (corn and soybean)	0.08	0.93	Sprayer	-	0.89
				Tractor	-	0.82
Paraná Use of legumes as cover crops (BR-2)	Spraying cover crops	0.10	0.8	-	-	0.31
Mecklenburg Use of cover crops (DE-2)	Seeding cover crop	-	-	Seeder (cultivator)	2.99	0.83
Mecklenburg Use of cover crops (DE-2)	Residue incorporation	0.14	6.5	Disk harrow	-	1.42
				Tractor	-	0.24

Note: Combination entails the sum of the changes of each individual strategy assessed in the Typical Farm, thus it is omitted from the table.
Source: own calculation based on focus groups.

Table A.14 Coefficients and resulting soil carbon stocks in the Typical Farm in Iowa

Strategy	Scenario	Reference carbon stock (SOC_{ref}) (t of C/ha)	Coefficients: Land use (C_{use}) -	Tillage intensity ($C_{tillage}$) -	Input of organic matter (C_{input}) -	Total carbon content (SOC_{total}) (t of C/ha)
Baseline	-	81	0.7	1.04	1	**58.97**
			(Long-term cultivated)	(Reduced)	(Medium)	
Reduce tillage intensity	Standard	81	0.7	1.09	1	**61.80**
			(Long-term cultivated)	(No-till)	(Medium)	
	Best	81	0.7	1.13	1	**64.28**
			(Long-term cultivated)	(No-till)	(Medium)	
	Worst	81	0.7	1.05	1	**59.33**
			(Long-term cultivated)	(No-till)	(Medium)	
Winter cover crops	Standard	81	0.7	1.04	1.11	**65.45**
			(Long-term cultivated)	(Reduced)	(High without manure)	
	Best	81	0.7	1.04	1.21	**71.99**
			(Long-term cultivated)	(Reduced)	(High without manure)	
	Worst	81	0.7	1.04	1	**58.97**
			(Long-term cultivated)	(Reduced)	(High without manure)	
Combination	Standard	81	0.7	1.09	1.11	**68.60**
			(Long-term cultivated)	(No-till)	(High without manure)	
	Best	81	0.7	1.13	1.21	**78.47**
			(Long-term cultivated)	(No-till)	(High without manure)	
	Worst	81	0.7	1.05	1	**59.33**
			(Long-term cultivated)	(No-till)	(High without manure)	

Note: Terminology in parentheses represent the denominations used by IPCC. To transform C to CO_{2eq}, the C value must be multiplied by 3.67.
Source: own estimation based on IPCC (2019a).

Table A.15 Coefficients and resulting soil carbon stocks in the Typical Farm in Mecklenburg-Vorpommern

Strategy	Scenario	Reference carbon stock (SOC_{ref}) (t of C/ha)	Coefficients: Land use (C_{use}) -	Coefficients: Tillage intensity ($C_{tillage}$) -	Coefficients: Input of organic matter (C_{input}) -	Total carbon content (SOC_{total}) (t of C/ha)
Baseline	-	81	0.7 (Long-term cultivated)	1.04 (Reduced)	1.00 (Medium)	**58.97**
Winter cover crops	Standard	81	0.7 (Long-term cultivated)	1.04 (Reduced)	1.11 (High without manure)	**65.45**
	Best	81	0.7 (Long-term cultivated)	1.04 (Reduced)	1.21 (High without manure)	**71.99**
	Worst	81	0.7 (Long-term cultivated)	1.04 (Reduced)	1.00 (High without manure)	**58.97**

Note: Terminology in parentheses represent the denominations used by IPCC. To transform C to CO_{2eq}, the C value must be multiplied by 3.67.
Source: own estimation based on IPCC (2019a).

Appendix B:
accompanying tables and results from the GHG mitigation strategies

Table B.1 Foregone revenue from yield penalty from transitioning the Typical Farm in Iowa to no-till and in combination of strategies

Year	Corn			Soybean			Average
	Yield penalty (%)	Yield penalty (t/ha)	Foregone revenue (USD/ha)	Yield penalty (%)	Yield penalty (t/ha)	Foregone revenue (USD/ha)	Foregone revenue (USD/ha)
1	6.0	0.76	97.6	1.8	0.07	22.8	60.2
2	5.7	0.72	90.9	1.7	0.07	21.2	56.1
3	5.4	0.69	84.4	1.6	0.06	19.7	52.1
4	5.1	0.65	78.2	1.5	0.06	18.2	48.2
5	4.8	0.61	72.1	1.4	0.05	16.8	44.5
6	4.5	0.57	66.3	1.4	0.05	15.5	40.9
7	4.2	0.53	60.7	1.3	0.05	14.2	37.4
8	3.9	0.50	55.2	1.2	0.04	12.9	34.1
9	3.6	0.46	50.0	1.1	0.04	11.7	30.8
10	3.3	0.42	44.9	1.0	0.04	10.5	27.7
11	3.0	0.38	40.0	0.9	0.03	9.3	24.7
12	2.7	0.34	35.3	0.8	0.03	8.2	21.8
13	2.4	0.30	30.8	0.7	0.03	7.2	19.0
14	2.1	0.27	26.4	0.6	0.02	6.2	16.3
15	1.8	0.23	22.2	0.5	0.02	5.2	13.7
16	1.5	0.19	18.1	0.5	0.02	4.2	11.2
17	1.2	0.15	14.2	0.4	0.01	3.3	8.8
18	0.9	0.11	10.5	0.3	0.01	2.4	6.4
19	0.6	0.08	6.8	0.2	0.01	1.6	4.2
20	0.3	0.04	3.4	0.1	0.00	0.8	2.1
Total	-	**-**	**908.2**	-	-	**211.8**	**560**
Annuity	-	**-**	**55.5**	-	-	**13**	**34.2**

Note: Initial yield of corn is 12.7 t/ha with a price of 130.7 USD/t. For soybeans, the initial yield is 3.8 t/h at 338.6 USD/t. The discount rate used is 2% and is already considered in the foregone revenue.
Source: own estimation.

Table B.2 Calculation of GHG emissions from corn and soybean from Indirect Land Use Change in Typical Farm in Iowa

Crop	EPA's indirect land use change (g CO_{2eq}/MJ)	Density of fuel (kg fuel/m^3)	Energy density (MJ/kg fuel)	Crop's biofuel yield (l/kg crop)	Emissions indirect land use (kg CO_{2eq}/kg crop)
Corn / Ethanol	39	794	27	0.3	**0.25**
Soybean / Biodiesel	56	890	37	0.2	**0.35**

Note: In the literature, the crop's biofuel yield is provided as a function of the dry matter. The values shown here are adjusted using a moisture content of 15.5% for corn and 13% for soybeans. These are the standard values for grain moisture content used in the Typical Farm in Iowa. EPA's Indirect Land Use Change values are expressed for the 20-year period.
Source:own estimation based on EPA (2010), Chen et al. (2018), Canter et al. (2016) and EU (2015).

Table B.3 **Total GHG emissions from Indirect Land Use Change from yield penalty from transitioning the Typical Farm in Iowa to no-till and in combination of strategies**

Year	Corn			Soybean			Average
	Yield penalty (%)	Yield penalty (t/ha)	Emissions (kg CO_{2eq}/ha)	Yield penalty (%)	Yield penalty (t/ha)	Emissions (kg CO_{2eq}/ha)	Emissions (kg CO_{2eq}/ha)
1	6.0	0.76	192.4	1.8	0.07	23.8	108.1
2	5.7	0.72	182.8	1.7	0.07	22.6	102.7
3	5.4	0.69	173.2	1.6	0.06	21.4	97.3
4	5.1	0.65	163.6	1.5	0.06	20.2	91.9
5	4.8	0.61	154.0	1.4	0.05	19.1	86.5
6	4.5	0.57	144.3	1.4	0.05	17.9	81.1
7	4.2	0.53	134.7	1.3	0.05	16.7	75.7
8	3.9	0.50	125.1	1.2	0.04	15.5	70.3
9	3.6	0.46	115.5	1.1	0.04	14.3	64.9
10	3.3	0.42	105.8	1.0	0.04	13.1	59.5
11	3.0	0.38	96.2	0.9	0.03	11.9	54.1
12	2.7	0.34	86.6	0.8	0.03	10.7	48.7
13	2.4	0.30	77.0	0.7	0.03	9.5	43.3
14	2.1	0.27	67.4	0.6	0.02	8.3	37.8
15	1.8	0.23	57.7	0.5	0.02	7.1	32.4
16	1.5	0.19	48.1	0.5	0.02	6.0	27.0
17	1.2	0.15	38.5	0.4	0.01	4.8	21.6
18	0.9	0.11	28.9	0.3	0.01	3.6	16.2
19	0.6	0.08	19.2	0.2	0.01	2.4	10.8
20	0.3	0.04	9.6	0.1	0.00	1.2	5.4
Average	**-**	**-**	**101**	**-**	**-**	**12.5**	**56.8**

Note: Initial yield of corn is 12.7 t/ha and of soybeans is 3.8 t/h. The emissions from Indirect Land Use Change are 0.3 kg CO_{2eq}/kg of corn and 0.4 kg CO_{2eq}/kg of soybeans.
Source: own estimation.

Table B.4 **Effect of nitrification inhibitor and nitrogen fertilizer reduction in combination of strategies in Typical Farm in Iowa**

Case	Emissions from direct			Leaching			Total fertilizer reduction
	N loss (kg N/ha)	Share reduced (%)	Fertilizer reduction (kg N/ha)	N loss (kg N/ha)	Share reduced (%)	Fertilizer reduction (kg N/ha)	(kg N/ha)
Standard	3.0	51	1.5	19.5	25	4.9	**6.4**
Best	2.8	55	1.5	11.1	25	2.8	**4.3**
Worst	3.3	47	1.5	27.7	25	6.9	**8.5**

Source: own estimation.

Table B.5 Effect of urease inhibitor and nitrogen fertilizer reduction in combination of strategies in Typical Farm in Paraná

Case	Urea rate (kg N/ha)	Losses as ammonia (kg N/ha)	Share reduced (%)	Urea reduction (kg N/ha)
Standard	138	5	47	2
Best	122	3	58	1
Worst	189	8	42	3

Source: own estimation.