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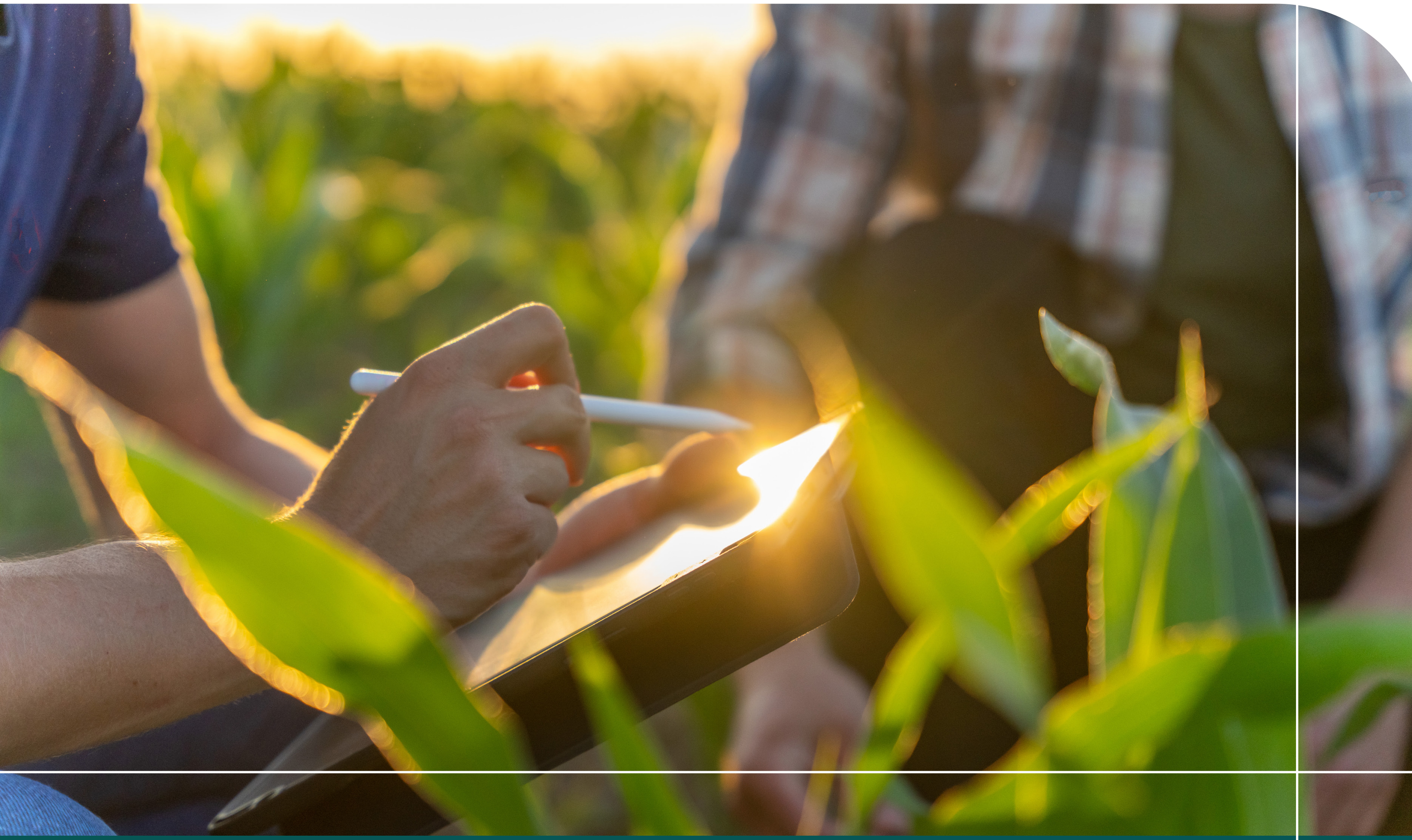

November 2024 | Full Report

Potential for U.S. Agriculture to Be Greenhouse Gas Negative

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In 2020, U.S. Farmers & Ranchers in Action (USFRA) established an independent scientific working group to analyze the potential for U.S. agriculture to collectively reduce greenhouse gas (GHG) emissions, including the potential to achieve a state of negative emissions, or emitting fewer total GHGs than are sequestered.

Building on a 2019 report by the National Academy of Sciences, Engineering and Medicine titled "Science Breakthroughs to Advance Food and Agricultural Research by 2030," the independent authoring group established by USFRA, consisting of 26 leading research scientists, identified current practices and emerging technologies with the most potential for reducing emissions. Their findings are based on a comprehensive analysis of scientific literature, computer simulations, and life cycle analysis estimates.

At USFRA's request and with support from the Foundation for Food & Agriculture Research, the National Academy of Sciences appointed a six-person committee to review the report, assessing its clarity, organizational effectiveness, and scientific rigor.

The final report, "Potential for U.S. Agriculture to Be Greenhouse Gas Negative," outlines how combining reduced GHG emissions from some agricultural activities with increased carbon sequestration in others could achieve GHG-negative agriculture. It also describes the research needed to help accomplish this.

We commend the members of the independent authoring group and National Academy of Sciences review committee for their commitment and substantial volunteer efforts throughout this multiyear endeavor.

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Chapter 1: Defining the Need for Achieving Greenhouse Gas Negative Agriculture

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Agriculture's Impact on Climate Change Globally and in the U.S.

The challenges facing humanity as we enter the middle of the 21st century are significant. The likely population expansion to 10 billion people by the end of the century will drive demand for the Earth's limited resources to levels never experienced by humanity. Estimates of future demand for food are dependent upon socio-economic as well as ecological conditions, but they range from a 50-100 percent increase in 50 years (Bodirsky et al., 2015). This is not a new challenge; over the past 10,000 years, human activities have transformed the Earth's approximately 130 million km^2 of ice-free surface (Ellis and Kaplan, 2013). This is the result of the dramatic success of our species to adapt and thrive under stressful conditions. However, the magnitude and rate of our exponential growth are new to the human experience; these extractive demands are stressing the biosphere to levels that may disrupt critical life support functions. Climate change resulting from the extraction of geologic carbon for energy is already altering weather patterns globally and may undermine the resiliency of humanity's food systems. We have transformed ecosystems, altered the hydrosphere, and are changing Earth's climate through emissions of greenhouse gasses (GHGs), predominantly carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) (Figure 1, after Smith et al., 2019). The total net annual anthropogenic contribution was 49 Gt CO_2-eq in 2016 (IPCC, 2023) and grew to 59 (+-6.6) Gt CO_2-eq by 2019 (Crippa et al., 2021; IPCC, 2023). In order to reduce net global warming to 2° C humanity will need to reduce net annual emissions of long-lived GHGs (CO_2 and N_2O) to near zero annually while reducing methane emissions from all sources (IPCC, 2023). Addressing these challenges requires a global perspective and meta-systems strategy that is adaptive and flexible. Total anthropogenic GHG emissions vary by category of source (sector) and location and are changing rapidly as emerging economies adopt new technologies, expand production and increase consumption of energy-dependent products (Crippa et al., 2020). The major GHG emitting sectors are energy (34 %), industry (24 %), net agriculture, forestry, and other land uses (AFOLU, 22%), transportation (15 %), and buildings (commercial and residential) (6 %), (IPCC, 2023). The distribution of net annual GHG emissions by the economic sector in the U.S. differs somewhat from the global emissions. The estimated total gross emissions from the U.S. in 2022 were 6.3 Gt CO_2-eq, or 5.5 Gt CO_2-eq, after land sequestration (USEPA, 2024). The U.S. contributed approximately 9.3 % of global GHG emissions in 2022. Transportation led the U.S. sectors at 29 %, followed by electricity (26 %), industry (22 %), commercial/residential (14 %), and agriculture (10 %). Total U.S. agriculture net GHG emissions were approximately 0.55 Gt CO_2-eq, or less than one percent of annual global GHG emissions in 2022 (USEPA, 2024).

Agricultural production is the most land-extensive human endeavor, occupying over 50 percent of the land on Earth. Agricultural land uses are categorized as cropland (14 %), pasture (21 %), and grazing on savannahs and scrubland (16 %) (IPCC, 2019). This extensive and complex global food supply chain can be characterized as a metasystem, defined as a series of systems within systems, forming an intricate and interconnected network. This metasystem exhibits both resilient and fragile characteristics across the global scale. The fragility of agricultural global food supply chain metasystems is amplified at the local scale.

The global food supply chain metasystem (the global food system) is remarkably efficient and complex, delivering nearly 22 trillion kilocalories of food to people across approximately 130 million km2 daily. The supply chain necessary to achieve this logistic miracle is equally complex and fragile. Market forces have largely shaped these supply chains and recent economic growth in emerging economies has accelerated global food distribution globally. Despite the efficiency and effectiveness of this global supply chain, the number of food insecure people has been rising since 2014. Currently, 690 million people (8.9% of the global population) are still hungry. Most (381 million) are in Asia, while Africa represents the fastest-growing number of undernourished people (250 million) (Bai et al., 2021; FAO et al., 2020). Even more alarming, two billion people (25% of the global population) were food insecure (did not have access to safe, nutritious, and sufficient nutrition) in 2019. In 2022 an estimated 45 million children under 5 years of age were suffering from wasting (UNICEF, 2023). The median cost of a nutritious diet globally was $3.75 per day in 2020, or $1,370 per person per year. A family of four would need to earn nearly $5,500 per year just for food. In the U.S. this number is almost three times the global level at $10 per person per day (Rabbitt et al., 2023). This is the economic threshold for the economic viability of a family and must be front of mind when considering other impacts of food production in the U.S. and globally.

The projected global grain yield for 2024 was 2.85 billion metric tonnes, rising consistently for the past decade (FAO, 2023). Global food production continues to be robust, with localized disruptions for short periods, even under the disruptions of the global COVID-19 pandemic (FAO, 2020). Global wheat and rice production experienced declines in 2020-2023 but stocks have remained constant. Meat production has expanded across all regions. Global food production systems are particularly vulnerable to extreme weather events and socio-political disruptions (FAO, 2023).

Agricultural production is a technological co-option of the global carbon cycle; plants transform atmospheric carbon dioxide (CO_2) into high-energy molecules (carbohydrates, protein, and fat). The inputs necessary to achieve the remarkable yields from the land are energy (and thus GHG) intensive. Net cumulative emissions for a region or globally are the total annual emissions minus the annual sequestration. The boundaries and assumptions applied in estimating global net GHG emissions can result in wide variability in the estimations of emissions. Net cumulative emissions of global anthropogenic GHG from agriculture for the 57-year period from 1961 – 2017 were 657 Gt CO_2-eq (range of 465-744 Gt CO_2-eq) while total annual anthropogenic GHG emissions from agriculture in 2017 were estimated to be 14.6 (range of 9.8-16.1) Gt CO_2-eq. (Hong et al., 2021) Crippa et al. (2021) used the EDGAR-FOOD database to estimate the global food system, including the full life cycle impact, to be 18 Gt CO_2-eq (95 percent confidence interval of 14-22 Gt CO_2-eq).

These ranges in estimated GHG emissions highlight the complexity of analyzing net GHG emissions from agriculture. Results can vary based on the systems boundaries and functional unit (unit of analysis). Many global boundaries include forestry (AFOLU), while others include agricultural land use and land use change (LULUC). Sector-level categories (often called product groups) are often compared, but product group assessments should always be interpreted with caution because the land use classes included in one assessment may not be the same as those in another, and the process of classifying land cover and land use within the U.S. let alone globally still has significant uncertainty. For example, cereals are responsible for the highest fraction of global total land use GHG emissions (47.4%), while chicken is only responsible for 0.6% (Hong et al., 2021). This does not mean all humans should only eat chicken. While this statement seems ridiculous, these are precisely the types of conclusions drawn from product group level analyses. Chickens require cereal crops for feed, while Bovidae (sheep, goats, and cattle) consume cellulosic materials that humans cannot digest. The more logical functional unit in a life cycle comparative analysis of agricultural impacts would be nutritional value for humans. This impact category includes metrics such as GHG emissions per calorie, nutritional density, critical nutrient (fat, protein, and carbohydrate) contribution, and other value characteristics to the human diet. This illustrates the complexity of the metasystems that compose human food supply chains.

Geospatial scales are critical as well. Global annual emissions are useful for benchmarking and tracking but not for analyzing, understanding, and reducing GHG emissions. The total proportion of GHG emissions from the food system has decreased 10% since 1990 (Crippa et al., 2021), but proportions are not measures of total impacts. This decrease was driven by reduced deforestation and increased agricultural efficiencies in emerging economies, resulting in real decreases in net GHG emissions in some agricultural sectors. One challenge for emerging economy agricultural production is the rapid increase of electricity and other energy sources inside the farm gate. These are discussed in Chapter 6 and are critical for balancing increased yields, efficiency, and reduced GHG emissions. The inter-relationship between emerging renewable energy sources and agricultural GHG emissions is clear. On-farm production of renewable energy saves the producer money, increases resiliency of the enterprise, and could contribute to reduced GHG emissions. Everything is connected.

What is Greenhouse Gas Negative Agriculture?

For the purposes of this report the term "Greenhouse Gas Negative Agriculture" means agricultural practices that emit less total greenhouse gasses than they sequester. This is a mass balance approach rather than a benchmark-reduction approach. The more technically accurate phrase would be " Net Negative Emissions Global Warming Potential Agriculture" but this phrase is cumbersome, so we adopted the shorthand "Greenhouse Gas Negative Agriculture" phrase. There are several methods for representing human impacts on global processes that impact solar insolation of the atmosphere (greenhouse gas effect). Global Warming Potential (GWP) is commonly used in Life Cycle Impact Assessments and by the IPCC and others (IPCC, 2023). The value of GWP is that it directly accounts for the radiative forcing consequences of specific GHG molecules (Derwent, 2020). For example, CH_4 has a GWP of about 28 over a 100-year time span (GWP100) (IPCC, 2023). This means that over the life of the molecule in the atmosphere, 1 Tg of CH_4 has the equivalent radiative forcing consequences in the atmosphere as 28 Tg of CO_2. This is the basis of CO_2-equivelent emissions calculations. The time frame is critical, however. The estimated lifetime of CH_4 in the atmosphere is only 10 years and is much shorter in warmer, humid zones (Derwent, 2020). Since CH_4 is continually emitted to the atmosphere and degrading; the net flux is integrated over the time of concern (usually 20, 50, and 100 years). Inventories of net GHG emissions are generally annualized to simplify benchmarking and sector-level performance. The major criticism of GWP as an indicator of climate change impact is that it is not tied to a net global temperature change but is a systems state analysis. To remedy this concern the Absolute Global Temperature Change Potential (AGTCP), in degrees K, is often presented along with GWP (Skytt et al., 2020). The Inventory of U.S. Greenhouse Gas Emissions and Sinks uses GWP100 for year-to-year comparisons of net GHG emissions by sector (USEPA, 2024). For this report, we will use net GHG emissions (CO_2-equivelent) based on

GWP100 unless otherwise indicated. The GWP timeframe is particularly important in animal agriculture, where CH_4 is a significant GHG contributor (see Chapter 5).

Greenhouse gas negative agriculture means that the net sequestration of greenhouse gas molecules is greater than the emissions from the agricultural sector. Globally the agriculture sector would have to sequester between 14 and 18 Gt CO_2-eq out of the total net emissions of 59 Gt CO_2-eq to be GHG negative by global sector. For the U.S. agricultural sector, total net GHG sequestration would have to reach approximately 0.60 Gt CO_2-eq to offset all emissions from U.S. agricultural production systems.

During the 21st Conference of the Parties to the United Nations Framework Convention on Climate Change in Paris (COP21) the Paris Climate Agreement was formalized as a strategy to limit global warming to less than 2° C. The global threshold for annual GHG emissions that would prevent exceeding the 2° C impact was estimated to be 9.8 Gt CO_2-eq, a more than 43 Gt CO_2-eq reduction from 2015 baseline emissions at the time, and almost 50 Gt CO_2-eq less than 2023 emission levels. If global net emissions from all AFOLU impacts reached zero (a reduction of global net GHG emissions of 22%, or 13 Gt CO_2-eq), we would still need to reduce global net GHG emissions another 37 Gt CO_2-eq to achieve the Paris Climate Agreement goal. Achieving zero emissions in agriculture is not ambitious enough to meet the Paris Climate Agreement goal, but it is a necessary step in achieving this goal. This report analyzes the potential for U.S. agriculture to achieve net negative GHG emissions, recognizing that global agricultural sector strategies will emerge from each country and hoping this analysis could be beneficial as a model for a pathway to carbon neutral agriculture,

The potential for soil systems to sequester carbon (the integration of organic carbon into soil carbon pools, predominantly soil organic carbon) has been recognized as a mitigation strategy for climate change for decades (Lal, 2004). Sequestering enough carbon in soil to offset all anthropogenic emissions might to be plausible with adequate innovation and adoption of key practices around the world (Sadatshojaei et al., 2021) but the economic costs for adoption of carbon sequestration practices could be too high. The potential for global soil systems to reduce net annual GHG emissions through sequestration would improve soil health by increasing soil organic carbon (SOC) and provide increased water holding capacities as well as other indicators of soil tilth resiliency (Minasy et al., 2017). Carbon net storage rates in soil are difficult to predict given the numerous drivers of carbon sequestration and storage (temperature, moisture, soil texture, historic soil management practices and current soil management practices, to name a few). Working Group III of the IPCC AR6 estimated with medium confidence that croplands could sequester between 0.4 and 6.8 Gt CO_2-eq. per year (IPCC, 2022). Thus, GHG-negative agriculture describes the process of net GHG sequestration in concert with reducing emissions across all agricultural production categories, including fuel, fertilizer, irrigation, cultivation, and other practices.

Categories of Agricultural Production Systems and Practices

The categories of agricultural production for this report are organized by process and production strategy. Process categories include soil carbon sequestration, nitrogen , water use efficiency, and on-farm energy use and production. Soil carbon sequestration is the key to GHG negative agriculture. Nitrogen use efficiency drives yield, reduces embodied GHG in the fertilizer and reduces N_2O emissions and water pollution. Water use is a major contributor to GHG emissions because irrigation requires energy to pump water from a source and distribute it to the crops. These three process categories are critical for reducing GHG emissions on crop lands, the major source of GHG emissions across all agricultural systems. On-farm energy use and production represent imminent innovations in carbon budgets for farm enterprises, as wind, solar, and biomass energy sources are developed to offset and replace petrochemical energy sources.

Production categories include row crop and animal protein production. These are closely linked processes because animals are fed row crops, and forage crops to convert low-density feeds into high-density and high-value foods, and animal manure is used to fertilize crops. Enteric methane production from animal agriculture is recognized as a significant source of CH_4 but the cumulative impacts on climate change are not well understood. Economic and social values integrate all production categories into a meta-systems-level decision framework.

The Purpose of this Report

The current state of knowledge regarding the practices representing the most substantial portfolio of choices for agricultural producers is incomplete. The drivers and barriers to adoption of those practices are not well documented. This report aims to analyze the potential of U.S. agricultural production supply chains to sequester carbon and reduce emissions. The approach was to identify the state of knowledge regarding the benefits and practicality implementation of specific practices necessary to achieve negative GHG emissions in agriculture and to create a roadmap for implementing the most promising strategies to achieve GHG negative agriculture.

This work builds on the National Academies of Sciences report "Science Breakthroughs to advance food and agricultural research by 2030" (NASEM, 2019). The challenges, opportunities, and gaps identified in the NASEM (2019) report are the starting point for the following discussions. The recommendations from this report are a subset of the broad findings of the previous work and provide a prioritized roadmap for research and implementation. While this report focuses on agricultural processes at the farm gate, the recommendations in Chapters seven through nine of the NASEM (2019) report are critical for creating negative GHG agriculture from producer to consumer.

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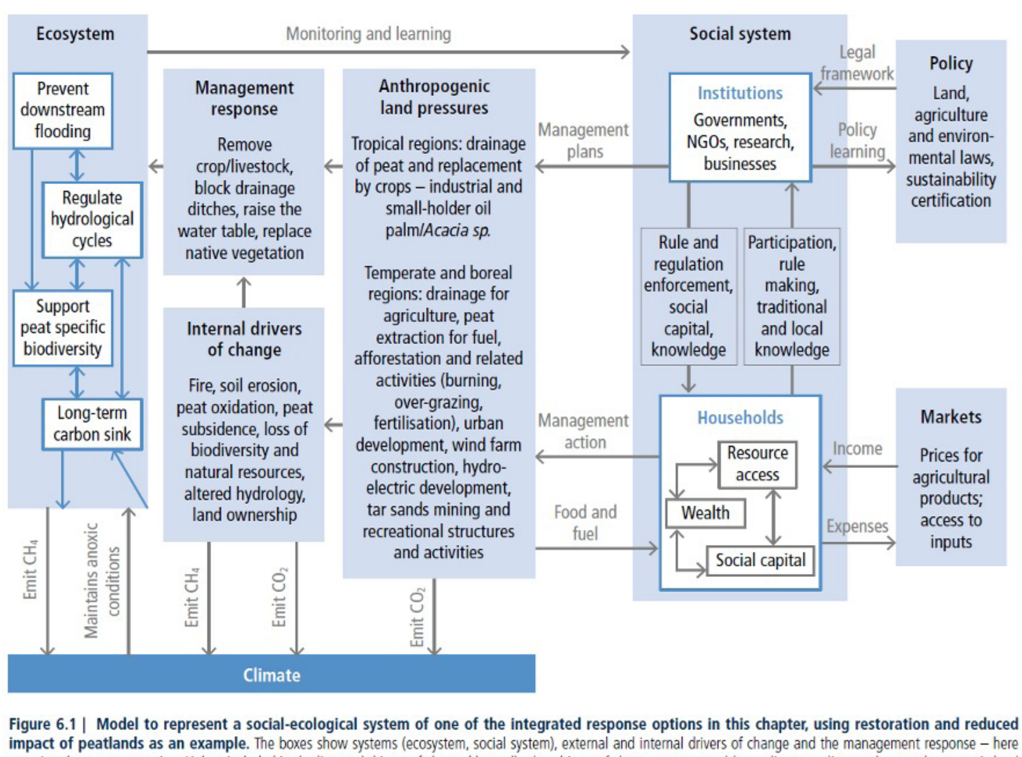

Chapter 2: The Challenges and Opportunities for Soil Carbon Sequestration

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Impetus for Pursuing Agricultural Management as a Negative-Emissions Strategy

Nearly 50% of the planet's land surface area is dedicated to agricultural systems, whether for crop production or pasture (FAO, 2020). Soil cultivation and grazing mismanagement has depleted global soil organic carbon (SOC) stocks in the top two meters of soil by as much as 133 petagrams (Pg), or 8% of total global SOC stocks (Sanderman et al.,2017). Land use conversion, agricultural intensification, and erosion have contributed to the historic loss of SOC, and these activities continue to deplete SOC in some regions. For example, as grasslands and forests are converted to croplands, more than 30% of native SOC is lost to the atmosphere (Poeplau et al., 2011). Historic mismanagement of croplands, particularly through intensive tillage and bare fallow practices, contributed to significant soil erosion (averaging 1.9 mm yr^{-1} in Midwest US) (Thaler et al., 2022) and SOC loss. In areas where conservation practices are underutilized, soil erosion and SOC loss continue at unsustainable rates. Coincident global change pressures (i.e., climate change, growing populations, urban encroachment, changing diets, demand for biofuels, and soil degradation) demand more from soil systems and lead to further SOC loss in many areas (Smith et al., 2016).

Restoring SOC to depleted agricultural soils and protecting existing SOC stocks can contribute substantially to atmospheric CO_2 drawdown and efforts to stabilize the climate system (Paustian et al., 2016; Griscom et al., 2017). Climate stabilization requires both drastically reducing fossil fuel C emissions and anthropogenic emissions of non-CO_2 greenhouse gases (CH_4, N_2O) and implementing measures to remove excess CO_2 in the atmosphere. Soil C sequestration, or the photosynthetic drawdown of CO_2 from the atmosphere and subsequent storage as soil organic matter, is recognized as a viable negative-emissions strategy. Even the most optimistic emissions reduction scenarios include substantial contributions from negative C technologies to keep below 2°C of warming (NASEM, 2019). This chapter will discuss the potential for agricultural best management to contribute to climate mitigation through soil C sequestration and storage in crop and grazing lands.

While soil C sequestration has become a major area of research focus in the realms of research and policy, studies on the relationship between C cycling in agricultural soils and climate mitigation began in the early 1990s (Barnwell et al., 1992; Paustian et al., 1997). The impact of soil and crop management practices on soil organic matter maintenance has been of interest since soil science and agronomy emerged as scientific disciplines in the early 19th century. Scientific inquiry into the link between soil organic matter and fertility goes back to the mid-18th century (Feller et al., 2014). Many long-term field experiments (LTEs) have tracked crop yields, nutrient dynamics, and soil organic matter changes as a function of crop rotation, tillage, nutrient management and irrigation over the past several decades (Leigh & Johnston, 1994; Paul et al., 1996). The oldest field experiments still in operation, at Rothamsted, UK, date back to 1843. Data from LTEs have facilitated the development of empirical models used in the Tier 1 methodology for national reporting of greenhouse gas (GHG) emissions from soils developed by the Intergovernmental Panel on Climate Change (IPCC) (Ogle et al., 2005; IPCC, 2006). Data from LTEs are also used to establish parameters for and validate dynamic process-based models that simulate soil C and soil GHG emissions as a function of management and environmental drivers (e.g., Campbell and Paustian, 2015; Basso et al., 2018). More recently, ecosystem-scale CO_2 flux measurements using eddy covariance methods have augmented the data sources available for testing and validating dynamic models (Zhang et al., 2018). Unfortunately, the U.S., unlike some other countries, does not have a measurement-based soil C stock inventory and monitoring system (van Wesemael et al., 2011), despite recommendations over a number of years by the scientific community (NRC, 2010; NASEM, 2019). Recently, USDA announced plans to evaluate the establishment of a national scale soil C and GHG monitoring network (USDA, 2023). However, current assessments of the biophysical potential for soil C sequestration rely chiefly on existing LTEs, eddy covariance sites, and systems modeling to determine the most effective strategies for increasing SOC stocks while minimizing non-CO_2 GHG emissions and maintaining crop yields.

Because of the documented potential to increase soil C stocks, along with a multitude of soil health and ecosystem co-benefits, managing agricultural soils has become part of the global agenda to mitigate climate change. Several global initiatives, including the International "4 per 1000" Initiative (Minasny et al., 2017), the Koronivia workshops on agriculture (UNFCCC, 2021), and FAO's RECSOIL program (FAO, 2019), all emphasize increasing SOC stocks as an important tool to draw down atmospheric CO_2 and increase resilience in the face of inevitable climate change. Beyond emerging national and international policies, the connection between soil and climate change has caught the attention of the private sector, from multinational corporations to smaller, environmentally conscious companies promoting market-based CO_2 mitigation. Private, voluntary C credit markets have a vested interest in soil C and recognize the untapped potential of agricultural projects to address the climate emergency, improve farmer livelihoods, and combat the degradation of arable lands worldwide. Additionally, popular documentaries like Kiss the Ground have brought the ideas of soil C and regenerative agriculture into homes across the country, perhaps with more optimism than is warranted (Amundson, 2021). Our role as scientists, and the purpose of this chapter, is to paint an accurate picture of how agricultural soil C sequestration can help mitigate climate change, without side-stepping the likely challenges.

Soil C Storage: Processes and limitations

The uppermost meter of global soils contains more C (1500 Pg) than the atmosphere (750 Pg) and terrestrial biomass (560 Pg) combined (Batjes, 2014). Because most cropland soils have lost 30-50% of native soil C (Davidson & Ackerman, 1993), agricultural soils have substantial potential to sequester C through improved soil management practices. Managing agricultural soils appropriately can accelerate the buildup of soil organic matter (SOM) by both controlling the type and frequency of organic matter additions and limiting soil disturbance that can stimulate organic matter decomposition. The amount of SOM in agricultural soils depends on a variety of climate, soil, and management factors, which vary seasonally and spatially. SOM forms through the progressive decay of plant and animal tissues, microbial biomass, manures, root exudates, and secondary compounds formed through decomposition. SOM has variable fates in the soil depending on the properties of the material, including chemical structure, C to nitrogen ratio (C:N), and solubility, as well as the properties of the surrounding soil environment. Mineralization of SOM, where organic inputs decompose or oxidize into plant available forms, is mainly driven by the synergistic activities of soil microbiota and larger soil fauna (e.g., earthworms) (Cotrufo et al., 2013; Liang et al., 2017). The soil food web and biogeochemical processes drive the progressive decomposition of SOM , as well as SOM respiration and loss.

Approximately 58% of SOM mass is C, with the remaining mass composed of hydrogen, oxygen, and other elements (e.g., nitrogen, phosphorus, and sulfur) essential to major biomolecules. Much of the C input into the soil mineralizes to CO_2 within a few years (Castellano et al., 2015). Additional C losses occur via soil system disturbance (e.g., tillage), leaching of dissolved organic C (DOC), and erosion (the latter two involve C translocation and deposition into a different location or into aquatic systems, not loss of C to the atmosphere). In general, soil C storage is guided by mass balance principles: when C inputs exceed C outputs, soil C stocks will increase, and the reverse. However, whether C additions are stabilized in the soil matrix for longer-term storage depends on many properties, such as quality of organic matter, soil type/texture and clay mineralogy, climate, and disturbance regimes. Generally, SOC accumulates more in cool, humid environments (where C inputs can be high, but cooler temperatures limit decomposition) than in hot and dry regions (Ogle et al., 2019). Additionally, soil C storage correlates with soil texture; fine-textured soils with more clay and/or silt tend to contain more SOC than coarse-textured soils.

Several proposed frameworks conceptualize the scientific understanding of SOM stabilization (Cotrufo et al., 2015; Lehmann & Kleber, 2015; Liang et al., 2017; Lehmann et al., 2020). Most research agrees that the efficiency of microbial processing of plant-derived residues, which depends on litter biochemistry, is the main control on SOM formation and stabilization over time (Cotrufo et al., 2013; Lehmann and Kleber, 2015; Liang et al., 2017; Lehmann et al., 2020; Liang & Zhu, 2021). Research supports microbial stabilization frameworks that show fungal and bacterial necromass (dead cells) are the primary constituents of stable SOM (Liang et al., 2019; Schweigert et al., 2015). Interactions between microbial byproducts, plant-litter derived compounds, and the soil mineral fraction determine the degree of SOM physical protection and persistence. At its simplest, SOM originating from microbial decomposition of organic residues can be divided into two pools with distinct pathways of formation and environmental persistence: 1) particulate organic matter (POM), composed of low density, OM fragments that are comprised of more chemically recalcitrant compounds often encapsulated within soil aggregates, or 2) mineral-associated organic matter (MAOM), composed of dissolved organic material and microbial byproducts that chemically bond with mineral particles (Lavallee et al., 2020) (Figure 1). The essential difference between MAOM and POM is the degree of stability: MAOM is protected from further microbial decomposition and the effects of system disturbance, while POM is more vulnerable. From a soil C sequestration perspective, management practices that increase MAOM stocks in soils should be prioritized because MAOM is more stable over time and less sensitive to disturbance (Haddix et al., 2020). On the other hand, MAOM is subject to saturation when as colloidal mineral surfaces become increasingly occupied. The saturation capacity of a given soil depends on the physiochemical characteristics inherent to the soil (e.g., soil texture and clay mineralogy) and the ecosystem in which the soil lies (Six et al., 2002; Stewart et al., 2007). Hence, ecosystem-specific management systems that support the formation of both MAOM and POM and prioritize the longevity of newly formed C stocks through reduced disturbance could effectively sequester C in soils and contribute to drawdown of atmospheric CO_2. An additional organic matter component in many soils, not derived from microbial-driven decomposition, is pyrogenic C (i.e., charcoal), formed from the burning of plant residues, or added as a produced organic amendment from pyrolysis of organic waste (Lehmann et al. 2006). Depending on the combustion conditions under which it was formed, pyrogenic C can be quite resistant to microbial decay and thus can be a persistent C fraction in the soil.

Management of Agricultural Lands for C Sequestration

Heterogeneity of organic matter inputs, microbial biomass and functional diversity, climate, soil type, time since disturbance, and management history, among other factors, drive soil C persistence, so the design of management systems to increase soil C stocks must holistically consider the context of the agroecosystem. Some agroecosystems are better suited for regeneration and C sequestration than others. Degraded soils that are substantially depleted of C have a high capacity, theoretically, to sequester C but also present the greatest challenges for regeneration if they have been degraded to the point that their primary productivity (and hence C input) has been significantly reduced (Chambers et al., 2016). The time necessary to recuperate lost soil C and improve agroecosystem functioning varies with climate, soil type, and historical and current management characteristics. Additionally, the unique C saturation potential and equilibrium point of the system limit SOC storage capacity (Stewart et al., 2007). There are no quick fixes for locking C away in soilsI. nstead, management should shift to a principle-based framework to improve the overall functioning of the agroecosystem, with the added benefit of improving C stocks (Paustian et al., 2016; Ogle et al., 2019).

Although the term has variable meanings depending on the context in which it is used, regenerative agriculture promotes a principles-based framework of agriculture with the goal of increasing and protecting soil C stocks, improving soil health, increasing profitability, and ensuring agricultural sustainability across the whole value chain (Newton et al., 2020). The following sections outline current best management practices for increasing soil C stocks and how these practices connect to the six principles of regenerative agriculture: 1) understanding agroecosystem context, 2) minimizing soil and ecological disturbance, 3) keeping soil covered, 4) maintaining living roots 5) maximizing diversity, 6) integrating animals Eckberg and Rosenzweig, 2021; see Figure 2).

1. Minimize disturbance: No-till, reduced tillage, and drainage management

Expanded no-till and reduced till practices in the Great Plains of the U.S. and similar agricultural regions around the world are due in part to the proven capacity of these practices to minimize erosion and prevent further soil degradation (Derpsch et al., 2010). No-till refers to the omission of plowing (soil disturbance) traditionally used for seedbed preparation and weed control. In no-till systems, crops are directly seeded into a field, allowing surface residues from the previous crop to protect against erosion and accelerated evaporation. No-till is often used to increase and maintain soil moisture in arid and semi-arid regions where rainfall limits crop yield (Palm et al., 2014). Additionally, no-till is a strategy to reduce atmospheric CO_2 because it protects existing SOC stocks by reducing soil disturbance, thereby increasing the mean residence time of SOC (cf. Six et al., 2000; Six and Paustian, 2014).

Results from nearly 200 field experiments assessing how no-till management affects soil C storage (Ogle et al., 2019) show a range of responses, from substantial C gains under no-till at some locations to reduced soil C stocks (relative to conventional tillage) at others. Several meta-analyses show that, on average, more C is stored in no-till soils than soils under full tillage management, with the most significant change in the top 20 cm of soil (Bai et al., 2019; Luo et al., 2010; Ogle et al., 2005, 2019; West & Post, 2002) (Table 1). However, the potential for C storage in agricultural systems under different tillage regimes depends on agroecosystem properties, including soil type, climate, and residue management (Ogle et al., 2019; Sun et al., 2020; Yuan et al., 2020). Changes in tillage regime may not always lead to an increase in C stocks (Powlson et al., 2014). Furthermore, increased C stocks in the topsoil may be partly offset by reduced C stocks at depth (Luo et al., 2010; Ogle et al., 2019). A literature review by Ogle et al. (2019) found no-till soils generally had higher SOC stocks in the surface soil (<20 cm), while full tillage soils often had higher SOC below the plow layer (>20 cm). Further analysis of C stock dynamics in the subsoil (and how these dynamics play out over time) is needed to explain how increases in SOC in the topsoil affect SOC at depth. Theoretically, as DOC and microbial byproducts are leached from the topsoil into the subsoil, the mineral-associated fraction will likely be enhanced, leading to higher long-term storage of stable SOC (Ogle et al., 2019).

Strategic deep tillage (where OM rich topsoil is transferred to the subsoil and OM-poor subsoil is brought to surface layers, and where plant residue inputs remain high) may increase total C storage in sandy soil in cool/temperate environments (Alcantara et al., 2016; Pereira et al., 2017). However, research must assess net C sequestration benefits despite the consequences of temporary decreases in topsoil fertility and increased fuel usage associated with deep tillage (Alcántara et al., 2016; Scanlan & Davies, 2019).

An additional source of soil disturbance is artificial drainage (e.g., tile drainage) which alters the hydrology of soil systems. Drainage systems are widely used to remove excess water from fields, encouraging soil aeration and allowing heavy machinery to access fields for cultivation and harvest. Removing standing water from fields decreases CH_4 and N_2O emissions associated with anoxic soil conditions but often leads to nutrients and DOC leaching out of the system (Ruark et al., 2009).

2. Maximize diversity: Crop rotations, microbial inoculum, and biochar

In unmanaged ecosystems, plant diversity increases soil microbial activity and soil C storage (Lange et al., 2015). However, the diversity of cropping systems has decreased globally in favor of monoculture systems that rely on synthetic inputs to manage fertility and pests. Diversifying crop rotations may be an effective strategy for decreasing reliance on inputs, improving the financial resilience of the production system, and increasing soil C stocks (McDaniel et al., 2014). In a global meta-analysis, McDaniel et al. (2014) found that adding one or more crops in rotation to a monoculture system increased total C stocks by 3.6%, on average, and microbial biomass C by 20.9%, yielding a mean SOC sequestration rate of 0.15 Mg C ha^{-1} yr^{-1} (Table 1). Additional diversity may be added to the system using perennial or prairie strips, which prevent erosion, reduce nutrient leaching into waterways, and support native pollinators (Schulte et al., 2017).

In addition to maximizing the floral and faunal diversity of agroecosystems, the diversity of C inputs to the soil can be enhanced directly by applying biochar amendments (Xu et al., 2021). Biochar is a C-rich soil amendment formed through the thermochemical conversion (pyrolysis) of a biomass feedstock, which renders the residual C less decomposable by soil microbes. Biochar amendments to agricultural soils can significantly affect soil C sequestration and storage by directly increasing the pool of persistent soil C and indirectly by enhancing soil aggregation and increasing system productivity through improved nutrient availability and soil water holding capacity (Du et al., 2017). Additionally, biochar may lead to a negative priming effect, where adding recalcitrant C decreases microbial decomposition of more labile C (Wang et al., 2016). Although this finding has yet to be fully confirmed, some positive priming effects have been reported. A recent meta-analysis found that applying biochar is among the most effective methods of increasing SOC content (up to 28% in field experiments), particularly over the short term (Bai et al., 2019). The costs of producing biochar, which is dependent upon the feedstock used, among other factors, may be prohibitive in some cases (Vochozka et al., 2016). In addition, the amount of waste biomass feedstock available for biochar production is a limiting factor (Schlesinger & Amundson, 2019). However, biochar may be an important tool to improve soil fertility and soil health, particularly in highly weathered acidic soils that are common in subtropical and tropical regions (Lehmann & Rondon, 2006). C

3. Keep the soil covered: Residue retention and cover crops

Keeping the soil covered provides many benefits to agricultural lands, including controlling erosion, moderating soil temperature, increasing water-holding capacity, and suppressing weeds (USDA NRCS, 2017). Soil cover can be maintained by leaving crop residues on the field, mulching, or planting cover crops in rotation with annual crops. From a soil system perspective, keeping the soil covered helps maintain soil C stocks by: 1) preventing wind and water erosion, which ensures C-rich topsoil remains in place; 2) moderating soil temperature and moisture loss, which prevents accelerated microbial respiration, and 3) increasing duration of vegetative cover, which increases overall C inputs.

Crop residues can be kept on the field after crop harvest to provide the soil cover benefits listed above and to maintain C inputs to the system that might otherwise be exported (e.g., corn stover harvested for feed, bedding, or biofuel feedstock). According to a meta-analysis of residue management studies, maintaining residue cover in no-till management systems is shown to increase SOC stocks more than no-till with residue removal (11% vs 13% increase) (Li et al., 2020). It is important to note that excess residue accumulation may interfere with no-till implements or cause other site-specific management challenges (Babcook, 2008).

Several meta-analyses and review papers have evaluated global data sets of cover cropping and associated impacts on SOC stocks and climate mitigation/adaptation (Jian et al., 2020; Kaye & Quemada, 2017; McClelland et al., 2021; Poeplau & Don, 2015). Poeplau and Don (2015) used linear regression of global cover crop data from 37 experiments to derive a response function for SOC stock change in the 0 to 30 cm soil layer. They found that SOC stock increased linearly with time after introducing cover crops, with an average change rate of 0.32 ± 0.08 Mg ha^{-1} yr^{-1}. In a more recent global meta-analysis of 131 studies, Jian et al. (2020) found a range of C sequestration rates from 0.22 to 0.71 Mg ha^{-1} yr^{-1} depending on soil texture, climate, and cropping system (Table 1). The meta-analysis also found that cover crop mixtures (particularly with legumes), as opposed to mono-species cover cropping, lead to greater increases in SOC (Jian et al., 2020). In another meta-analysis, cover crops were associated with an average SOC stock increase of 1.11 Mg ha^{-1} in the top 0 to 30 cm, compared to a no cover crop control (McClelland et al., 2021).

In some cases, experimental studies can show negative C stock changes after cover cropping or, alternatively, surprisingly high values of C accrual. Negative C stock changes may be due to the effects of priming,where adding low C:N crops leads to greater microbial respiration (Jian et al., 2020) or reduced productivity of the subsequent cash crop (Lobell and Villoria, 2023). The wide range in values of SOC stock change can be attributed to the heterogeneity of SOC, which is characteristic of all soil systems, along with initial C stocks, insufficient time since adopting cover crops, and sampling methods that fail to take this heterogeneity into account (Poeplau & Don, 2015).

4. Maintain living roots: Agroforestry and perennial crops

Increasing the perenniality of agricultural systems by maintaining above and below ground C inputs across space and time through cover crops and crop rotations is an effective strategy for increasing SOC in agricultural systems (King & Blesh, 2018). In recent models of SOM dynamics, root exudates and root biomass are hypothesized to be more important to the formation of stable SOM (in both the MAOM and POM fractions) than aboveground plant material (Cotrufo et al., 2013; Sokol et al., 2019). This is likely because root deposits can be directly absorbed to mineral surfaces and can bolster microbial biomass growth in the rhizosphere (McDaniel et al., 2014). The expansion of agroforestry and perennial staple crops offers an opportunity to greatly increase soil C sequestration in agricultural soils (Toensmeier, 2016). When we account for both above and belowground C sequestration, agroforestry systems often offer the most benefit per hectare C storage rates (De Stefano & Jacobson, 2018). While developing perennial grains may offer significant soil C and water use efficiency benefits (de Oliveira et al., 2020), the challenges of achieving economically viable yields (compared to conventional annual crops) should not be overlooked. Finally, protecting C stocks in existing native perennial grasslands, wetlands, and forests is essential to avoid further soil C losses from these terrestrial systems (De Stefano & Jacobson, 2018; Bossio et al., 2020).

5. Integrate livestock: Integrated crop & livestock systems and grazing management

While the primary focus of this chapter is cropland management for soil C storage, pastures and rangelands account for nearly 70% of global agricultural land and are of critical importance to C storage. Livestock grazing has the potential to increase soil C storage, but the appropriate grazing practices (e.g., rotation and intensity) to support C gains and avoid losses is dependent on climate, soil type, and other environmental factors. Prolonged high-intensity grazing (overgrazing) frequently decreases soil C stocks (Zhou et al., 2017), however, short-duration, intensive (rotational) grazing may increase soil C, particularly in moist, warm climates (Abdalla et al., 2019). In some dry environments, soil C stocks may increase by reducing grazing pressure (Zhou et al., 2017). However, soil C responses to grazing intensity can vary strongly as a function of grass species/ecotypes (Milchunas & Lauenroth, 1993). While grazing removes aboveground biomass and heavy grazing can decrease net primary productivity (NPP), grazing may also lead to increased fungal dominance belowground, contributing to more soil C storage and resilience to changing moisture regimes (Abdalla et al., 2019). In addition to controlled grazing intensity, pastures managed with adaptive multi-paddock grazing (AMP), or short duration, high stock density grazing rotations, had 13% more soil C (9 Mg C ha^{-1}) in the top 1 m of soil than neighboring conventionally grazed sites (Mosier et al., 2021).

Additionally, integrated livestock and annual cropping systems, where livestock graze cover crops or crop residues, may improve soil C storage and soil health while reducing system GHG emissions (Salton et al., 2014). Furthermore, grazing livestock on non-cash crops provides an additional revenue stream that may improve the financial resilience of the farm system, particularly as annual crop yields become less stable with climate change (Peterson et al., 2020). Particularly in semi-arid systems where C inputs from annual crops are low, crop rotations that include intermittent grazed forage crops increase C inputs to soil and accrual of SOC (Brewer and Gaudin, 2020). In the absence of grazing animals, animal manures may be broadcasted on or injected into cropland fields to derive some of the benefits of an integrated animal livestock system. Manure application provides fertility benefits that may otherwise be provided solely by synthetic fertilizers, generating use value out of an otherwise difficult to manage waste product. Manure application is shown to increase soil C stocks but associated increases in field GHG emissions (namely N_2O) must also be considered. A full accounting of the potential benefits of integrated crop livestock versus decoupled crop and livestock production requires a comprehensive life cycle analysis of all components of each system (Liebig et al., 2021).

6. Understanding context: Agroecosystem specificity and co-benefits

The sixth principle is not always included in descriptions of regenerative agriculture, but it is perhaps the most important. There are no one-size-fits-all solutions for soil management to maximize C storage; management must instead be adapted to unique soil types, climate, and production needs of the agroecosystem. Ideally, the combined effects of regenerative practices work synergistically to confer the greatest benefits to the soil system, leading to high rates of C accumulation (Bai et al., 2019). However, with so few established long-term experiments, relative to the diversity of potential systems, assessing the impact of many different combinations of regenerative practices is difficult. In the absence of established experiments, recent observational studies have worked in collaboration with dedicated producers to evaluate the impact of their regenerative management systems (e.g.,(LaCanne & Lundgren, 2018; Luján Soto et al., 2021; Mosier et al., 2021; van der Pol et al., 2022)) and meta-analyses provide information about the impact of integrating several best management practices simultaneously (Bai et al., 2019; Yuan et al., 2020).

The effects of regenerative practices should be viewed from a systems level, looking beyond CO_2 inputs and outputs to other GHGs. For example, producing and applying N-fertilizers is the primary source of N_2O emissions, a greenhouse gas with ~300 times the global warming potential of CO_2. Therefore, practices that increase SOC and reduce emissions from all GHGs sources across the agricultural value chain should be favored.

Reducing N_2O emissions from fertilizers is an important co-benefit of implementing regenerative practices that increase organic N availability by increasing SOM. Practices that maintain soil cover and living roots are another example of systems that increase agroecosystem resilience to changing and unpredictable climate conditions. Protecting bare soil reduces soil erosion, maintains C in topsoil, and insulates soil to reduce evapotranspiration and soil temperatures. Furthermore, increasing crop diversity protects against pathogens and disease outbreaks, represses undesirable weed species, and diversifies producer revenue streams. Once the initial costs of transition (e.g., new equipment, potential yield reductions) are recovered, regenerative agriculture has the potential to decrease production costs through reduced input costs, fewer passes with farm machinery, and less yield loss due to extreme weather and pests (Liu et al., 2018; Singh & Meena, 2013).

Biophysical potential for soil C sequestration/ negative emissions on U.S. cropland

As part of a recent U.S. national decarbonization analysis (Larson et al., 2020), we estimated the potential to increase soil C stocks on managed croplands in the U.S. A recent National Academies report (NASEM, 2019) classified soil C sequestration technologies into two main categories: existing conservation practices and frontier technologies. Frontier technologies include advances such as crop varieties with enhanced root phenotypes (i.e., larger, deeper root systems), perennial grains, widespread use of biochar amendments, and other technologies still in the research phase but not yet ready for widespread deployment in U.S. agricultural systems (NASEM, 2019). In contrast, existing conservation practices (described as regenerative earlier in this paper) are relatively well understood and have been deployed to varying degrees in production agriculture (e.g., cover crops, intensified rotations, no-till or reduced tillage, and integrated crop-livestock systems). U.S. cropland acres planted to cover crops increased 17% from 2017 to 2022, but this still only amounts to 4.7% of total cropland (USDA NASS, 2022). Conservation tillage practices were used on approximately 75% of corn/soybean, 68% of wheat, and 43% of cotton acres in the U.S. in 2021. Conservation tillage includes both no-till and mulch tillage (using a chisel plow or disk) (USDA NASS, 2022). Only half of the reported conservation tillage acres utilize no-till, and tillage practices may vary with point in the crop rotation (i.e., periodic tillage in corn/soybean systems) or management challenges (i.e., compaction or residue accumulation). For example, many corn/soybean acres in the Midwest are managed using periodic no-till, usually meaning no-till practices are used during soybean years while conventional tillage is used during corn years. Even a one-time tillage event may lead to the loss of 1-10% of SOC (Conant et al., 2007), negating the potential SOC storage benefits of conservation tillage and other complementary practices. The main challenge in implementing these existing practices for C drawdown is increasing their level of continuous adoption on the greatest number of acres possible.

For the current estimate of negative emissions potential, we considered only existing conservation management practices, not frontier technologies. While many studies report significant potential for increasing soil C stocks on grazing lands (e.g., Conant et al., 2017; Mosier et al., 2021), field data are sparser than for annual cropping systems, and current (baseline) grazing management systems are not well documented. Thus, our estimates here are limited to agricultural land presently used as annual cropland. The potential C drawdown from all agricultural lands, including grazing lands, and the potential for widespread adoption of new agronomic technologies could have significantly more impact than the results presented here.

Our estimates for the cropland soil C sink and potential for reducing net GHG emissions in the U.S. are derived from 1) annual croplands, 2) cropland purposed for bioenergy crop production systems, and 3) land set aside from crop production for conservation. To estimate potential agricultural land negative emissions in 2050, we considered the total land area available within each land use category, selecting one of two scenarios for bioenergy production developed for a broader cross-sectoral/cross-technology negative emissions assessment (see Larson et al., 2020). We chose the delimited bioenergy scenario, which included only the conversion of corn grain ethanol croplands to perennial biomass energy crops (6.9% of annual cropland area) and did not include any conversion of marginal croplands to biomass energy crops, as in the high bioenergy scenario (U.S. DOE, 2016). Two mitigation sub-scenarios (called 'moderate' and 'widespread' below, based on the percent of annual cropland converted to perennial grass conservation cover) were applied only on annual croplands not converted to perennial biomass energy crop production (Table 2).

Total baseline areas of cropland were extracted at the county-scale from the 2017 U.S. Agricultural Census (USDA-NASS, 2017). The agricultural land areas that could contribute to potential soil C stock increases were grouped into three categories: 1) land currently used for corn grain ethanol production that could be converted to perennial grass biomass crops, 2) marginal cropland area converted to perennial vegetation for conservation, and 3) cropland remaining in annual crop production using soil conservation management practices (Table 3). We conservatively estimated a small portion (5% and 10% in moderate and widespread adoption scenarios, respectively) of current annual cropland would be available to convert to permanent herbaceous cover for conservation within cropland, including field borders, filter strips, grass waterways, and riparian buffers. Recent estimates suggest that more than 20% of cropland in an average field in the Midwest has much lower productivity than other parts of the field and may even produce negative net revenues (Basso et al., 2019). Thus, consistently low productivity areas could be more profitably set aside as an in-field conservation area.

Once current and future agricultural land bases were established (Table 3), areas were determined for the adoption of moderate scenarios of conservation practices that could sequester atmospheric C and more widespread adoption scenarios using conservation practices that could reduce GHG emissions (see Table 2). Practices were chosen based on the potential to reduce emissions and the practical scalability of implementation (see Table 2). Emission reduction coefficients associated with adopting USDA-Natural Resource Conservation Service Conservation Practices Standards (USDA-NRCS, 2022) were derived from the COMET-Planner Tool (Swan et al., 2020). Values in COMET-Planner represent regionally-averaged soil C and GHG emissions computed with the DayCent biogeochemical simulation model within the COMET-Farm platform for completing field-scale C and GHG inventories (Paustian et al., 2018). The COMET-Planner tool reports net changes in soil C stocks and soil nitrous oxide emissions (as CO_2 equivalents) from implementing soil conservation management. Negative values indicate a net reduction of GHG emissions relative to baseline agricultural management. Emission reductions due to increases in soil C under consistent management should continue for approximately 20-30 years on average before approaching a new state of equilibrium (Paustian, 2014). Soil nitrous oxide emission reductions would continue indefinitely, assuming consistent management over time in baseline and conservation scenarios.

To avoid double counting emission reductions on lands currently practicing conservation management, we removed those land areas from future projections to the extent possible. The 2017 Agricultural Census provides data on the current use of no-till and cover crops but does not provide data on areas under both no-till and cover crops. Because no-till has been adopted on more acres than cover crops, we conservatively removed all land area already under no-till management from future projections. Similarly, we did not estimate future reductions in emissions for lands currently enrolled in the Conservation Reserve Program (CRP), which pays farmers to temporarily convert annual crop to perennial grass or tree cover.

Under the widespread conservation management adoption scenario, we estimated an overall net GHG emission decrease of approximately 234 million metric tons (MMT) CO2-eq yr^{-1} (Table 3). Across the U.S., we estimated an average per ha GHG emission decrease on current cropland area of 1.47 Mg CO2-eq ha^{-1} yr^{-1} relative to current agricultural management. With a moderate level of conservation practice adoption, we estimated a reduction in total net emissions of 133 MMT CO2-eq yr^{-1} , approximately 57% of the decrease projected for the widespread adoption level.

The geographic patterns of C sequestration and net GHG reductions on a per unit area basis generally reflect climate patterns in the U.S., with higher sequestration rates predicted for humid climates or irrigated systems (more than 2-3 Mg CO2-eq ha^{-1} yr^{-1}) and lower rates predicted in drier climates under rainfed conditions (< 1 Mg CO2-eq ha^{-1} yr^{-1}) (Figure 3). When applied to the area of agricultural lands, the highest potential for C sequestration and GHG emission reductions are in the rainfed (largely non-irrigated) croplands of the northern Great Plains, Midwest, and Mississippi Delta regions and irrigated croplands in the West (Figure 4).

According to our analysis, converting corn grain ethanol to perennial energy grasses and using perennial grass conservation set asides could reduce emissions by 36 MMT CO2-eq yr^{-1}, representing about 15% of total emission reductions under the high bioenergy/widespread conservation adoption scenario. Geographically, most conversion of corn grain ethanol to perennial energy grass would be in the Midwest, with smaller areas spread throughout the eastern U.S. (Figure 5). The potential for increased soil C sequestration following the conversion of annual crops (including corn) to perennial grass biomass feedstocks such as switchgrass is well established from field and model-based studies (Field et al., 2020).

Our estimates of the potential for removing CO_2 from the atmosphere using widespread adoption of conservation practices on U.S. croplands align with several other estimates ranging between 170 to 290 MMT CO2-eq yr^{-1} (Lal, 2003; Morgan et al., 2010; Sperow, 2016, 2020). In a recent analysis, Robertson et al. (2022) estimated CO_2 removal and GHG emission reduction for US annual cropland as well as land conversions to perennial biomass production for bioenergy. Based on empirical models derived from LTE data, they estimated potential SOC gains of 208 MMT CO2-eq yr^{-1} on annual cropland employing a similar suite of conservation practices for which we estimated 204 MMT CO2-eq yr^{-1}. Robertson et al. (2022) estimated an additional net emission reduction of 99 MMT CO2-eq yr^{-1} on annual cropland from N_2O and rice CH_4 reductions, and avoided CO_2 emissions from industrial fertilizer use and rewetting of peat soils, which were not included in our analysis. Their study also included a more in-depth look at the potential for perennial biomass feedstocks (e.g., switchgrass) on marginal lands and land used for corn ethanol production. They estimated a total SOC increase potential on available land (i.e., 10 million ha (Mha) currently used for corn ethanol, 5 Mha from Conservation Reserve Program land, 41 Mha of non-forested abandoned agricultural land), if converted to switchgrass, could yield an additional increase of 101 MMT CO2-eq yr^{-1}. Our total assumed cropland area available for conversion to perennial grass (24 Mha; Table 2) was much less and hence our total SOC gain from conversion on perennial grass (30 MMT CO2-eq yr^{-1}) was less.

The degree of convergence between our estimates and others recent estimates, including those above, is not surprising, given that the major determining factors (available cropland area and rates of soil C accrual after adopting conservation practices) are well defined and widely used by different analysts. The greatest variable in estimating potential C removal by cropland soils is estimating the proportion of total land area where conservation practices may be adopted. If low rates of adoption are assumed, then accordingly the potential for soil C sequestration will appear to be low. Adoption, of course, depends to a large extent on economic and policy conditions that may incentivize or inadvertently discourage farmers from adopting soil C sequestering practices, as well as increased outreach, training, and technological innovation.

The U.S. national greenhouse gas inventory (EPA, 2023) currently estimates an annual increase of 15 MMT CO2-eq yr^{-1} in baseline soil C stocks for cropland remaining under cropland management, with cropland converted to perennial grass (e.g., pasture) accounting for an additional soil C stock increase of 23 MMT CO2-eq yr^{-1}. Both would add up to almost 50 MMT CO2-eq yr^{-1}. These baseline increases are based on the current modest adoption rates of conservation practices, including reduced tillage and no-till, conservation plantings on marginal lands (i.e., CRP), field buffers, and grassed waterways. Thus, if more farmers adopt C sequestration practices (particularly cover crops), achieving an additional 100-200 MMT CO2-eq yr^{-1} is feasible. However, note that the U.S. GHG inventory also estimates a loss of about 50 MMT CO2-eq yr^{-1} from converting grassland and forest into annual cropland, effectively canceling out current estimated soil C gains for other land uses. Such soil C losses associated with land use change underline the importance of avoiding the conversion of perennial vegetation, including grasslands and forests, to cropland.

Concluding Remarks

The scientific community has progressed toward a robust understanding of the mechanisms controlling SOC stabilization and storage, as well as the key role of SOC in ecosystem services. Aggregate data from LTEs and biogeochemical modeling provide estimates of the potential for existing conservation management practices to reduce agricultural GHG emissions and contribute to drawdown of excess atmospheric CO_2 .

Without underselling the potential benefits of regenerative agriculture (stabilizing the climate system and improving agroecosystem functioning), we must remain aware of likely challenges. As demonstrated in our analysis, the reduction of potential emissions and removal of existing atmospheric CO_2 associated with management changes differ greatly across diverse ecoregions and soil conditions in the U.S. Furthermore, the potential impacts of climate change on soil C stocks will vary by region and will require region-specific management responses (i.e., shifts to warmer and/or wetter climates, intensified droughts, impacts of large precipitation events, etc.) Coordinated efforts to model county-level C sequestration potentials, accounting for soil type, climate, current/historic land use, and future climate scenarios should guide land use conversion and promote conservation management. Further research recommendations, including nation-wide measurements and monitoring, systems-level research, and technological developments are outlined in Table 4.

Although SOC was depleted from agricultural soils in a geologic instant, it will take time and coordinated effort to replenish C stocks across the range of generally C-depleted U.S. cropland soils. Biophysical/technical potential provides strong evidence that we can achieve GHG reductions and C drawdown of 100-200 MMT CO2-eq yr-1 (over two to three decades) on U.S. cropland if the adoption of regenerative management continues and is widespread. However, the socio-economic and political challenges associated with the change of this scale should not be downplayed. Effective economic incentives, technical guidance, and social support structures are needed to meet the urgency of this challenge.

With widespread adoption of regenerative management, cropland soils can contribute 17 to 33% of the 600 MMT CO2-eq yr^{-1} needed to offset U.S. agricultural emissions, as proposed in Chapter 1. Given the urgent need to stabilize the climate system, agricultural management to prevent further GHG emissions and increase soil C storage should be employed as one of many diverse strategies to mitigate atmospheric C and protect terrestrial C stores.

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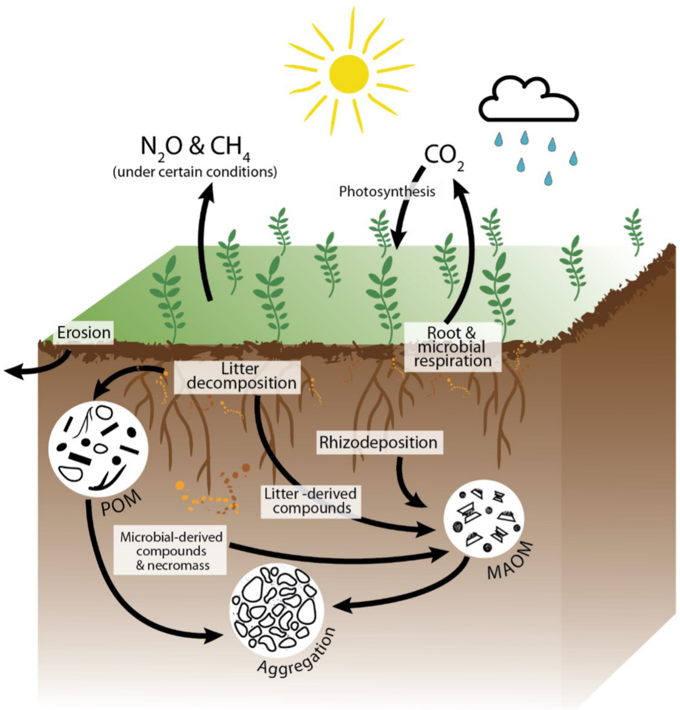

Figure 1. Schematic diagram of soil C cycling and stabilization mechanisms. Most SOC is lost as CO_2 through heterotrophic and autotrophic respiration. SOC can be stabilized through 1) biochemical stabilization via higher recalcitrance and 2) physical stabilization in organo-mineral complexes and aggregates.

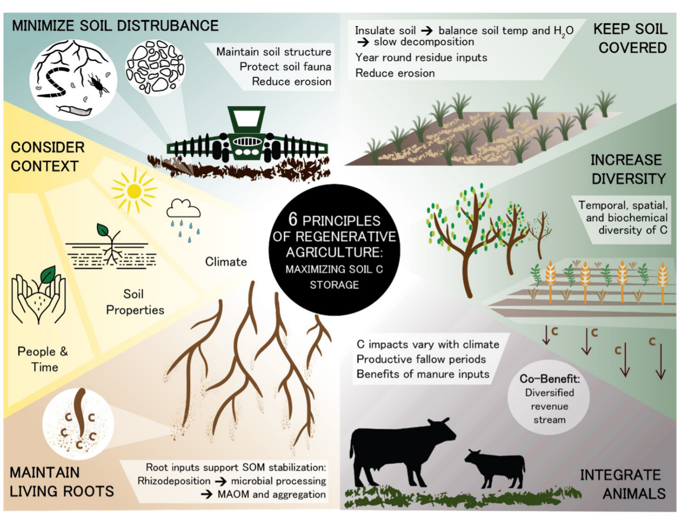

Figure 2. Diagram illustrating how the 6 principles of regenerative agriculture relate to soil C accumulation and stabilization mechanisms. The 6 principles of regenerative agriculture included here are 1) consider agroecosystem context, 2) minimize soil disturbance, 3) keep soil covered, 4) increase diversity, 5) maintain living roots, and 6) integrate animals. These principles can help maximize the C sequestration and general functioning of agroecosystems.

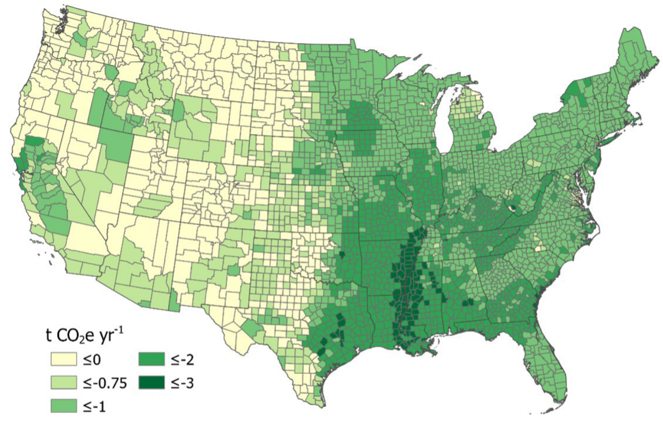

Figure 3. Per unit area rates of net negative emissions (in Mg CO_2-eq ha^{-1} yr^{-1}) for all cropland uses at the county scale for the widespread adoption scenario. Negative emissions are relative to the baseline emission/removals.

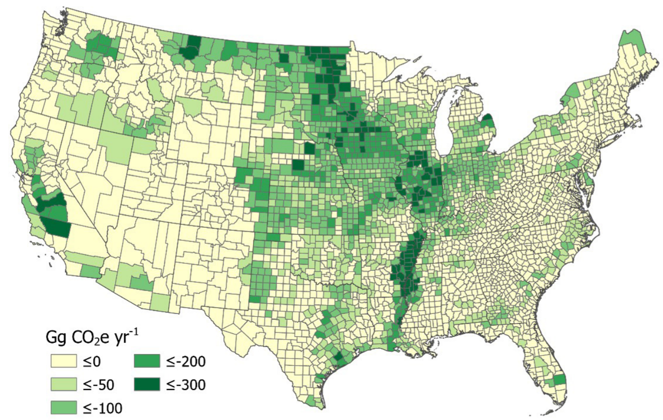

Figure 4. Net negative emissions at county-scale (in Gg CO_2-eq yr^{-1} or 103 Mg CO_2-eq yr-1) for all cropland uses at the county scale for the widespread adoption scenario. Negative emissions are relative to the baseline emission/removals.

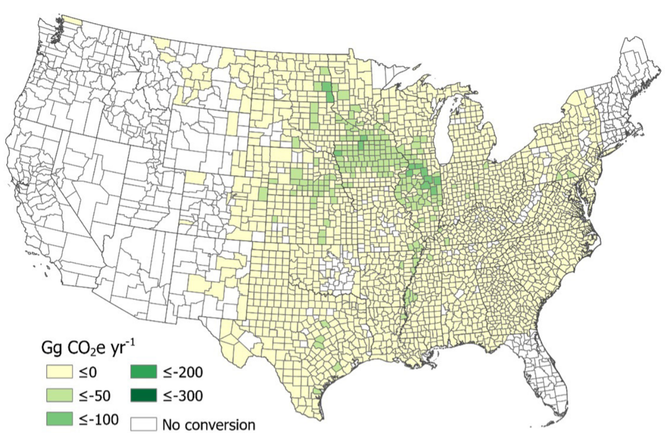

Figure 5. Total GHG changes (in Gg CO2-eq yr^{-1} or 103 Mg CO_2-eq yr^{-1}) due to conversion of corn grain ethanol to perennial energy grasses at the county scale for the widespread adoption scenario. Negative emissions are relative to the baseline emission/removals.

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Table 1. Meta-analyses reporting mean ΔSOC (C sequestration rate) in Mg C ha-1 yr-1 with mean values by climate type and soil texture (highest and lowest values reported), where available. * Our summary estimates assume 10 years of biochar application to derive an annual average increase.

Practice	Reference (Author, Year)	Overall mean ΔSOC (Mg ha-1 yr-1)	Mean ΔSOC by climate type (Mg ha-1 yr-1)	Mean ΔSOC by soil texture (Mg ha-1 yr-1)
Cover cropping	(Don and Poeplau, 2015)1(Jian et al., 2020)2	0.32[1] ; 0.56[2]	(Tropical - 0.71 ± 0.21; Arid 0.46 ± 0.22)2	(Fine - 0.82 ± 0.30; Coarse - 0.43 ± 0.12)[2]
No-till	(Ogle et al., 2019)1(Bai et al., 2019)2	0.38[2]	(Tropical - 0.34 to 0.54; Arid - 0.06 to 0.15)1	(Fine - 0.06 to 0.54; Coarse - 0.15 to 0.50)[1]
Diversified crop rotation	(McDaniel, Tiemann, and Grandy, 2014)	0.15	Not evaluated	Not evaluated
Biochar application	(Gross et al., 2021)	1.30*	(Temperate – 1.13; Tropical – 0.46)*	(Fine – 1.28 to 1.77; Coarse – 0.61)*
Manure application	(Gross and Glaser, 2021)	1.07*	(Non-tropical – 1.28; Tropical - 0.85)*	(Fine - 1.13 to 1.17; Coarse – 0.82)*

Table 2. Summary of mitigation scenarios by land use category and the USDA/NRCS Conservation Practice Standards (CPS) applied (USDA-NRCS, 2022). The two right columns are percent of total cropland area to which the standards are applied.

Land Use	USDA/NRCS Conservation Practice Standard	Moderate Adoption	Widespread Adoption
		Percent Land Area	
Croplands remaining under crops	Humid Climates: No-Till (CPS 329) + Cover Crops (CPS 340) Dry Climates: No-Till on Irrigated Croplands (CPS 329); Conservation Crop Rotation (CPS 328) on Non-Irrigated Croplands	44.4%	85%
Croplands Converted to Permanent Herbaceous Cover	Conservation Cover (CPS 327)	3.8%	8.1%
Corn Ethanol Area Converted to Biomass Energy Crops	Forage and Biomass Plantings (CPS 512)	6.9%	6.9%

Moderate Adoption Scenario	106 hectares	MMT CO2-eq/yr
Corn ethanol converted to herbaceous biomass crops	11	-23
Marginal cropland area converted to perennial cover	6	-4
Conservation practices on annual cropland	71	-106
Total	88	-133

Widespread Adoption Scenario	106 hectares	MMT CO2-eq/yr
Corn ethanol converted to herbaceous biomass crops	11	-23
Marginal cropland area converted to perennial cover	13	-7
Conservation practices on annual cropland	136	-204
Total	160	-234

Table 3. Total land areas affected by land use conversions and adoption of conservation management practices; resulting total net CO2-eq emission changes (as negative emissions).

Research need	Recommended Research Action	Description and Justification
Expanding monitoring & measurement	National on-farm monitoring system and integrated model data platform1,2,3,4	• Development of national system for measuring and modeling regional on-farm C sequestration accounting for soil type, climate, current/historic land use, and socio-economic factors impacting transition to negative C practices • National standards for sampling, analysis, modeling, and data reporting • Systems-level analysis of ecosystem services and effects beyond productivity, including emissions of all pertinent GHGs across the production system (ex. Sustainable Intensification Assessment Framework) • National assessment of alignment to and delivery on UN Sustainable Development Goals
Interdisciplinary systems-research	Agricultural systems field experiment network1,2,3	• Expand upon existing network of long-term agricultural experiments and collaborative on-farm research endeavors to evaluate region-specific best management practices and effects of integrating multiple practices simultaneously • Emphasize understanding C stock dynamics in the subsoil (below 30 cm) • Measure changes in SOM fractions, with implications for stability, permanence, and sensitivity to system disturbance. Effect of integrating multiple regenerative management practices simultaneously
	Socio-economic barriers to increasing landowner adoption of negative C practices, quantifying economic benefits of increasing SOC2,3	• Interdisciplinary research to understand socio-economic barriers to adopting negative C agriculture practices and opportunities for co-benefits • Research to inform development of economic and policy structures as incentives to adopting negative C practices • Co-development of knowledge with producers to inform research, understand barriers to adoption, and provide effective technical support and training
Frontier research and technology development	High through-put, low-cost methods to monitor changes in SOC stocks	• Reducing cost and time associated with soil C monitoring through robustly calibrated and verified technologies that measure SOC and bulk density in the field (e.g., advances in field spectroscopy, non-invasive bulk density measurement)
	High C input crop phenotypes and perennial staple crops2	• Developing high C input crop phenotypes with altered root morphology and biomass • Using perennial staple grain and oilseed crops • Analysis of the soil and ecosystem C effects of on-farm perennial strips, agroforestry systems, etc.
	Innovative soil amendments for improved C sequestration and soil function2	• Microbial inoculants, seed coatings, and genetic engineering for novel plant-microbe associations • Full LCA of biochar production system (including various feedstocks) and effects on nutrient cycling and non-CO2 GHG emissions

Table 4. Research recommendations for the development, expansion, and implementation of negative C agriculture practices, technologies, and monitoring (References: 1. Middendorf et al., 2020; 2. NASEM, 2019; 3. Rumpel et al., 2020; 4. Smith et al., 2021)

Chapter 3: The Challenges and Opportunities for Nitrogen Use Efficiency

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Introduction

Agriculture directly contributes ~12% of all global anthropogenic greenhouse gas emissions annually (IPCC 2019), with ~36% of these emissions directly from nitrous oxide (N_2O) emissions. Changes in land cover, although not part of agriculture's direct emissions, contribute another 11% of total global emissions, with a small portion derived from N_2O producers stimulated by land clearing. Nitrogen fertilizer production contributes another 1-2% of total global emissions. All told, agriculture contributes ~24% of global greenhouse gas emissions, and nitrogen, particularly N_2O production, is a significant part of that total.

At the farm scale, achieving net zero carbon emissions requires abating both N_2O emissions and N fertilizer use. Even considering the carbon cost of agronomic inputs like diesel, pesticides, and soil amendments like lime, nitrogen drives greenhouse gas balances. For example, in a long-term corn-soybean-wheat rotation in the upper U.S. Midwest, N_2O emissions contributed 35% and nitrogen fertilizer 33% of the system's total global warming impact of 98 g CO_2e per m_2 yr^{-1} (Gelfand et al., 2013). Fuel, seeds, pesticides, lime, and other fertilizer inputs made up the remaining 32%.

It stands to reason, then, that agriculture has significant opportunities for mitigating climate change through proper nitrogen management. Avoiding N_2O emissions saves the climate from further atmospheric N_2O loading. Avoiding nitrogen fertilizer saves the carbon cost of fertilizer manufacture. Better management of nitrogen in cropping systems thus has a significant mitigation payoff.

Nitrous Oxide Emissions

The significance of N_2O comes not from its atmospheric mass, which is vanishingly small at ca. 330 parts per billion by volume, but from its high global warming potential (GWP), which is a measure of the capacity of a well-mixed gas to trap heat once emitted to the atmosphere (IPCC, 1990). By definition, GWP is relative to a reference gas, by convention CO_2, and its magnitude depends on the efficiency with which the gas absorbs long-wave radiation (heat), its existing mass in the atmosphere, and its atmospheric lifetime; e.g., long-lived gases like N_2O (with its 109 year atmospheric lifetime) will have high GWPs even if their heat trapping efficiency is relatively low. While imperfect, GWP is one of several useful metrics for calculating the relative effects of different greenhouse gases on radiative forcing (IPCC 2021).

N_2O has a 100-year GWP of 273 (IPCC 2021), which means that it traps heat in the atmosphere ~300 times more effectively than CO_2. Thus, a kilogram of N_2O emitted to the atmosphere is equivalent to ~300 kg of CO_2 emitted at the same time. Conversely, from a mitigation standpoint, avoiding emission of 1 kg of N_2O is equivalent to sequestering ~300 kg of CO_2 with the added advantage that in avoiding N_2O emission, the gas is not at risk of being re-emitted at a later date, as is the case with sequestered soil carbon. Thus, among agricultural strategies to mitigate greenhouse gas emissions, N_2O abatement has a relatively high payoff.

Atmospheric N_2O concentrations are rising at a rate of ~2% per decade, an accelerated pace compared to 50 years ago, largely because of more intense agricultural production. Best estimates place the current global flux for anthropogenic activities at ~7.3 Tg N_2O-N yr^{-1} (Tian et al. 2020). About 80% of this flux is attributed to agriculture: 50% to cropland soils and the downstream ecosystems to which they contribute excess nitrogen, 17% to soils in grazed pastures, and the remainder to a combination of manure management in confined feeding operations, biomass burning in the tropics, and aquaculture. The remaining global anthropogenic flux comprises fossil fuel combustion and industrial sources such as nitric acid production (see Figure 1).

Biological sources

Biotic sources of N_2O dominate the global cycle, with denitrification and nitrification the primary microbial pathways by which N_2O is produced. Both processes occur readily in soil when nitrogen is available and environmental conditions like moisture and temperature are favorable (Robertson and Groffman 2021). During nitrification, ammonium (NH_4+) added as fertilizer or mineralized from soil organic matter, crop residues, or other added organic materials (including manure) is oxidized to nitrite (NO_2-) and eventually to nitrate (NO_3-) in a series of reactions that can also produce N_2O:

[Eq. 1]

$$NH_4^+ \longrightarrow \overset{\overset{N_2O}{\uparrow}}{NH_2OH} \longrightarrow \overset{\overset{N_2O}{\uparrow}}{NO_2^-} \longrightarrow NO_3^-$$

Dentrifiers use nitrate and other oxidized forms of nitrogen as electron acceptors during cellular respiration when oxygen is in limited supply. In denitrification, N_2O is an intermediate product that can either escape to the atmosphere or be further reduced to N_2 gas, thereby helping to close the global nitrogen cycle:

[Eq. 2]

$$2NO_3^- \longrightarrow 2NO_2^- \longrightarrow 2NO \longrightarrow N_2O \longrightarrow N_2$$

Nitrifiers can also denitrify, so distinguishing how much N_2O is exclusively due to nitrification is difficult. However, both isotopomer studies (e.g., Ostrom et al. 2010, Buchen et al. 2018) and recent whole-soil kinetic analyses (Liang and Robertson 2021) suggest that N_2O from nitrification, where it occurs, is likely to be a minor direct source of N_2O, at least in the ecosystems thus far examined. That said, nitrification is almost always the major source of nitrate in agricultural soils (nitrate fertilizer is uncommon, and rainfall usually adds little), so nitrification as a compulsory precursor to denitrification can have a controlling influence. This control is especially evident in wetland systems like lowland rice where anaerobic soil conditions suppress nitrification and consequently can suppress denitrification.

Theoretically, atmospheric N_2O can also diffuse to denitrifiers and be reduced directly to N_2. However, because net positive fluxes tend to be much more common and larger than net negative fluxes (Chapuis-Lardy et al. 2007), soils are a negligible global sink for N_2O (Schlesinger 2009).

To recap, soil N_2O fluxes are largely controlled by environmental factors that control nitrification and denitrification: soil temperature and moisture as they affect microbial activity in general and as moisture affects the diffusion of oxygen to microsites where nitrification and denitrification occur; soil carbon availability as it affects the availability of electron donors to denitrifiers and the consumption of oxygen in soil microsites; and especially the availability of soil inorganic nitrogen (NH_4+ for nitrifiers, NO_3- for denitrifiers).

Soil inorganic nitrogen is derived from the mineralization of soil organic matter, from added nitrogen fertilizers, either synthetic or organic, and from grazing animals. A relatively small amount, typically <10 kg N ha^{-1} annually, but sometimes more in heavily polluted regions, enters agricultural systems via precipitation. Soil structure plays an important role in N_2O production in that sandy or otherwise poorly structured soils have fewer anaerobic microsites than well-structured or heavier soils with higher clay contents. Therefore, sandier soils typically have lower rates of denitrification although this may matter little in soils with other types of anaerobic microsites such as those inside soil aggregates (Hojberg et al. 1994), within detritus particles (Kravchenko et al. 2017), and during periods when soil is saturated such as occur during wintertime thaws (Ruan and Robertson 2017).

In situ factors that affect N_2O production are many and vary across spatial and temporal scales (Figure 2). The large number of factors that affect nitrification and denitrification at fine temporal scales, from hours to weeks, help explain the episodic nature of soil N_2O emissions (e.g., Barton et al. 2015, Grace et al. 2020), with more distal, slower-acting controls such as cropping system and topographic position providing the envelope within which the finer scales operate.

Soil nitrogen as master control

The importance of soil nitrogen availability for controlling N_2O emissions in agricultural soils is related to the ability of added nitrogen, whether in synthetic or organic form, to stimulate both nitrification and denitrification. At the field scale, soil nitrogen availability is the single best predictor of N_2O emissions (e.g., Gelfand et al. 2016, Saha et al. 2021), which is why IPCC national greenhouse gas inventories are by default based on a simple percentage of nitrogen inputs (IPCC 2006).

However, the relationship between nitrogen fertilizer inputs and N_2O emissions is more complex than the simple linear relationship used by the IPCC, as revealed by recent field experiments. In fertilizer response trials, when nitrogen fertilizer is added to crops at different rates to define both yield and N_2O emission responses to added nitrogen, yields typically increase to some point after which additional nitrogen has no effect (Figure 2a). Nitrous oxide fluxes, on the other hand, typically remain low until crop yields equilibrate and then rise exponentially (e.g., Hoben et al. 2011, Millar et al. 2018; see Figure 2b), presumably because after yields level off, nitrogen not used by the crop is available for microbial uptake. In ecological terms, the microbes are released from competition with plants for available nitrogen. This exponential relationship, by now well established but not universal, applies globally in a wide range of cropping systems (Shcherbak et al. 2014).

The underlying processes that produce N_2O have a complex suite of controlling factors, including abiotic controls like soil moisture, temperature, carbon, oxygen, and pH (Robertson and Groffman 2021), and biotic factors such as population-level differences in nitrogen uptake kinetics (Liang and Robertson 2021) and sensitivities to oxygen (Cavigelli and Robertson 2001). Nitrogen availability, however, remains the master control. In almost all soils, added nitrogen stimulates N_2O emissions, and the withdrawal of nitrogen through, for example, plant uptake or microbial immobilization suppresses emissions.

Agricultural management and N_2O emissions

Soil nitrogen availability most often limits soil N_2O emissions, so any agronomic practice that increases soil inorganic nitrogen concentrations will likely also accelerate N_2O emissions. Fertilizing with anhydrous ammonia, urea ammonium nitrate, or any of the inorganic fertilizer formulations is the most obvious practice that will elevate soil inorganic nitrogen pools, but even without adding synthetic fertilizer, crop and grazing lands management can increase soil inorganic nitrogen availability. This occurs most intentionally via nitrogen fixing crops such as legumes (including soybeans, alfalfa, and clover), by adding manure and compost, and following tillage, which helps decompose crop residue, especially when leguminous cover crops are incorporated before planting a cash crop. Inorganic nitrogen pools are elevated unintentionally following fall harvest and when soils are left bare during the non-growing season. Residual nitrogen left over from growing season additions and nitrogen added by newly decomposing crop residues can increase soil inorganic nitrogen pools substantially. If temperature and other conditions like low oxygen availability due to high soil water contents are favorable for denitrifiers, N_2O production can be substantial even during the winter (Ruan and Robertson 2017, Wagner-Riddle et al. 2017).

In general, therefore, abating N_2O emissions requires soil management that avoids excess nitrogen availability, i.e., any level of soil nitrogen that exceeds existing plant needs. The first line of defense, then, is fertilization rates. A second important option is growing perennial crops or keeping continuous green cover through the judicious use of cover crops (Mosier et al. 2021). Relative to annual crops, perennial non-leguminous crops such as various forage grasses or bioenergy feedstocks like switchgrass (Panicum virgatum) or poplar trees (Populus spp.) tend to emit N_2O at very low rates, especially when unfertilized (Oates et al. 2016). Because such crops are actively growing during a greater proportion of the growing season, have deep persistent roots that can capture and store residual soil nitrogen, have lower nitrogen needs in general, and avoid the need for annual tillage as well, soil inorganic nitrogen levels tend to be lower in systems with perennial crops. The exceptions are perennial legume crops with their high capacity for biological nitrogen fixation (Robertson and Groffman 2021); for example, N_2O emissions from alfalfa stands can be as high as emissions from fertilized annual crops (Gelfand et al. 2016).

Some nitrogen conserving benefits of perennial crops can also be achieved with cover crops planted at or before annual crop harvest. Especially if non-leguminous, cover crops like annual rye grass (Lolium multiflorum L.) or sorghum sudan grass (Sorghum bicolor x S. bicolor var. Sudanese) can capture residual fertilizer nitrogen and the nitrogen newly mineralized from decomposing crop residues, keeping inorganic nitrogen away from the soil microbes that produce N_2O and keeping nitrate from being leached from the system to become available for N_2O production downstream.

Other nitrogen management strategies can also help to minimize N_2O emissions. Fertilizer management based on 4R principles (Bruulsema et al. 2012), right timing, placement, and formulation in addition to right rate, can also improve nitrogen conservation. Timing fertilizer application to when crop need is highest, for example, can minimize microbe exposure. About one-third of Midwest corn crops received anhydrous ammonia in the fall, 6-8 months before crop need, effectively pre-fertilizing the microbes. Applying this at planting, or better, as a side-dress application several weeks after planting, can conserve nitrogen that would otherwise be emitted as N_2O over winter and spring or leached downstream for later conversion to N_2O (Ogle et al. 2014). More advanced timing technologies such as on-the-go sensors deployed throughout the growing season on specialized tractors or through pumped irrigation systems can add nitrogen at even more precisely timed intervals, as can slow-release fertilizers.

Placement can also be important. Adding fertilizer closer to growing plant roots by injecting liquid fertilizer within the crop row or using near-row drip lines can also improve nitrogen conservation, as can using precision technologies to fertilize different parts of a field at rates that better match plant nitrogen needs.

Different nitrogen fertilizer formulations affect N_2O emissions in many systems. Anhydrous ammonia in the U.S., for example, is the most common form of nitrogen fertilizer applied to field crops. It stimulates N_2O production with rates 40 to 200% higher than broadcast urea at the same nitrogen rate (Ogle et al. 2014). Fertilizers can also be formulated to enhance uptake efficiency. Such fertilizer formulations include polymer-coated urea to delay nitrogen release until temperature and moisture conditions favor plant growth; stabilizers, or urease inhibitors, to delay the hydrolysis of urea-nitrogen to ammonium; and nitrification inhibitors to delay the microbial conversion of ammonium to nitrate. All these products can make nitrogen supply more synchronous with plant nitrogen demand and thereby reduce fertilizer nitrogen need and subsequent nitrogen loss.

That said, improved nitrogen use efficiency depends on nitrogen fertilizer rate: only when nitrogen rates are lower than optimal do advanced formulations improve plant nitrogen access and yields (Rose et al. 2018), which helps explain the inconsistent N_2O effects found in many studies (Akiyama et al. 2010, Halvorson et al. 2014, Hatfield and Venterea 2014). N_2O abatement benefits are likely the result of reducing nitrogen rates, not a direct effect on microbial processes producing N_2O. Inhibitors also face the challenge of seasonal persistence. Improving synchrony during the growing season is not enough; N_2O is also produced after the growing season when unused nitrogen and nitrogen newly mineralized from crop residue become available to soil nitrifiers and denitrifiers.

Finally, tillage can affect cropland N_2O emissions. By stimulating nitrogen mineralization, redistributing carbon, impairing soil structure, and reducing infiltration rates, tillage can create the carbon, nitrogen, and oxygen conditions favoring denitrifiers. On the other hand, no-till management can also favor anaerobic microenvironments for denitrifiers, especially inside soil aggregates (Sexstone et al. 1985) and in detritus particles adjacent to soil pores (Kravchenko et al. 2017). These contrasting effects help to explain results from long-term studies that suggest no consistent differences in N_2O emissions between conventional and no-till systems for the first decade following no-till establishment, although after 10 years, emissions show a net reduction, especially in drier environments (Van Kessel et al. 2013). As for yield and other potential benefits of no-till, N_2O abatement can take years to be consistently expressed (Cusser et al. 2020).

Nitrogen use efficiency

Nitrogen use efficiency (NUE) is a broadly used term to denote the physiological, agronomic, and environmental efficiencies with which nitrogen is used in ecosystems. Ladha et al. (2005) detailed at least 18 ways to calculate NUE, and from the standpoint of nitrogen conservation in general and N_2O abatement in particular, system-wide NUE provides the most relevant metric for evaluating the likelihood of excess nitrogen loss (Robertson and Vitousek 2009). Most system-wide NUE metrics are calculated as the balance between nitrogen added and harvest removals. For systems with the same yield, those with more nitrogen removed at harvest than added from fertilizer and other sources during the growing season have a higher NUE and thus are more nitrogen conservative.

The system-wide NUE of major cereal crops is less than 50% globally: 42% for wheat, 39% for rice, and 46% for maize (Lassaletta et al. 2014, Udvardi et al. 2021), which means that less than half of added nitrogen from all sources (fertilizer, biological nitrogen fixation, and the decomposition of crop residues) is taken up by the crop. The remainder is lost to the environment unless the soil-crop system is storing nitrogen internally, which would be unusual for well-equilibrated cropping systems not accumulating soil organic matter. Fruits, vegetables, nuts, and other high value crops like seed corn have substantially lower system-level NUEs because farmers, often motivated by suppliers (Stuart and Houser 2018), tend to use even more fertilizer despite the fact that so much is wasted (Udvardi et al. 2021). Unfertilized legumes are the exception. For crops that rely exclusively on biological nitrogen fixation, such as soybeans, system-wide NUE can be as high as 80% (Córdova et al. 2019).

Regardless of the source of nitrogen, N_2O emissions are high wherever excess nitrogen is available and the other environmental factors that affect microbes that produce N_2O are favorable. Thus, as noted earlier, N_2O production in alfalfa, soybean, and other leguminous field crops can be as high as N_2O production in fertilized cereal crops, despite legumes' relatively high system-wide NUE as long as excess nitrogen is available when moisture, temperature, and labile soil carbon are present. Because agroecosystems have been intentionally designed to provide crops with ample nitrogen (in natural ecosystems the nutrient most often in shortest supply), it is not surprising that N_2O emissions will be high without explicit efforts to abate them.

New Opportunities For N_2O Abatement

Strategies for abating N_2O emissions in cropland agriculture are frustratingly few. No direct inhibitors work consistently in most soils. No single management factor, other than nitrogen fertilizer rate, can unambiguously suppress N_2O production. Instead, field-scale NUE using a suite of management interventions is necessary to keep excess soil nitrogen from becoming atmospheric N_2O. The interventions already noted, more precise fertilizer rates, cover crops, biological nitrogen fixation, advanced fertilizer formulations, no-till, and fertilizer timing and placement to maximize plant-soil synchrony, are together important and must be incorporated into systems appropriately to maintain current yields; otherwise, new crop production elsewhere to make up market shortfalls will create new greenhouse gas emissions that negate intended benefits.

Fortunately, such interventions also make cropping systems more resilient, regenerative, and profitable, improving soil quality and reducing agriculture's environmental impact (Sherwood and Uphoff 2000, Robertson et al. 2014, Spiegal et al. 2018, Giller et al. 2021). Co-benefits abound, including climate benefits other than N_2O mitigation. Reduced nitrogen fertilizer use saves much of the carbon cost of fertilizer manufacture (Gelfand and Robertson 2015), and cover crops both build soil organic matter (Poeplau and Don 2015), which sequesters soil carbon (Paustian et al., this volume) and increase off-season albedo, leading to net climate cooling (Dominique et al. 2018). To reap the full synergistic benefits of integrated solutions requires a systems approach to management (Swinton et al. 2007), and digital agriculture (see Basso and Antle 2020) is particularly promising for achieving several sustainability objectives simultaneously, including greenhouse gas mitigation.

Digital agriculture's promise for mitigating N_2O emissions

Agriculture is in the midst of a digital revolution. As in manufacturing, agriculture has begun using integrated smart technology for increased automation, improved self-monitoring, and the capacity to analyze, diagnose, and communicate production issues without human intervention. Digital agriculture, designing and adopting smart technologies to collect, manage, and apply data to improve the efficiency of agricultural operations, can lead to more sustainable, resilient, and circular agricultural systems through the efficient use of human and agricultural resources. At the field scale, digital agriculture can integrate spatial and temporal variability to balance inputs and management interventions. Factors that affect crop yield, like soil water and nutrient availability, plant density, and pest and disease pressure, vary substantially across individual fields (Robertson et al. 1997, Maestrini and Basso 2018), and new sensors ranging in scale from individual plants to satellites allow collecting timely information to manage field scale variability at high spatial and temporal resolutions.

System-wide NUE was an early target of digital agriculture. NUE is spatially variable at sub-hectare scales in most cropping systems (e.g., Mamo et al. 2003, Scharf et al. 2005), a consequence of varying soil properties, microclimates, and pest and plant populations. Better matching nitrogen supply with plant needs should be readily achievable using plant sensors that can detect nitrogen status and yield as well as farm equipment that can apply nitrogen fertilizer at different rates across individual fields (Maestrini and Basso 2018). In the U.S., ~60% of farmers use global positioning systems (GPS) to guide field operations, 68% have yield monitor sensors, but fewer than 20% have variable rate equipment, which is mostly used to plant seeds and apply pesticides and fertilizers other than nitrogen (Lowenberg-Deboer and Erickson 2019).

One explanation of the slow adoption rate for variable rate nitrogen technology is the difficulty of accurately assessing causes of variability. For instance, plants are assumed to be nitrogen limited in subfield areas with low yields or low soil nitrogen instead of available water or some other factor, which can lead to over-fertilization and even lower NUE, with concomitant economic and environmental harm. Because most intensively cropped soils in the U.S. are over- rather than under-fertilized with nitrogen, the main advantage of variable rate fertilizer technology is to avoid overfertilizing subfield areas where plants cannot use available nitrogen because of other growth constraints. Process-based crop simulation models can help avoid such mistakes but are not widely deployed. Such models, when they are geospatially explicit, can identify which areas in a field are likely responsive to additional nitrogen, which allows inputs tailored to plant growth potentials.

As an alternative to soil tests, process-based geospatial modeling, or on-the-go plant analysis, historical yield patterns can consistently identify low performing subfield areas that can be targeted for NUE intervention. Multi-year analysis of spatial and temporal variation of crop yields obtained from sensors mounted on harvesters (Basso et al. 2007, Maestrini and Basso 2018) or from satellite images (Basso et al. 2019) can reveal subfield areas with consistently high yields, consistently low yields, and varying yields. For example, Basso et al. (2019) examined subfield yields for six years across 29 million ha in the U.S. Midwest to show that 48% of within field variability is characterized by stable high yields, 27% by stable low yields, and the rest by yields that were high in some years and low in others. The system wide NUE for different subfield stability classes, calculated as the difference between nitrogen inputs and harvest output, varied from an average of 80% in consistently high yield areas to 45% in areas of stable low yields. The stable low yield areas were responsible for the bulk of regional excess nitrogen losses. Northrup et al. (2021) extended these findings to suggest that applying spatially variable N fertilizer based on yield stability maps could reduce fertilizer application by 36% and N_2O emissions by 23%.

Another strategy for using digital agriculture to mitigate N_2O emissions is to convert consistently low yielding areas to conservation plantings (Basso 2021) or perennial bioenergy crops. Apart from substantially lower N_2O emissions, co-benefits include soil carbon sequestration, higher water quality downstream, and biodiversity, which can also benefit yields (Nelson and Burchfield 2021). Either of these strategies, variable rate nitrogen delivery and precision conservation, can substantially reduce N_2O emissions from cropped fields (Figure 4).

Research Needs

Reducing N_2O emissions in agriculture requires new research to address three principle challenges:

1. Improve field-scale N use efficiency

Soil N availability is the master variable for predicting and controlling N_2O emissions, and in agricultural systems, reducing soil N availability without affecting yield requires increasing NUE at multiple scales, ranging from plant-microbe interactions to fertilizer technology (Udvardi et al., 2021). At the plant scale, genomic advances can help in designing and deploying root systems that can better extract N from soil. Better understanding of the plant-soil microbiome could improve N acquisition and stress tolerance as well as encourage associative N fixing microbes in the rhizosphere to reduce reliance on N fertilizers and their embedded CO_2 costs. At the field scale, developing fertilizer technologies that release N mainly or only when there are actively growing roots could vastly improve NUE. In addition, precision fertilizer management, applying fertilizer based on where and when it is needed instead of applying fertilizer at one rate once or twice per year, could help improve NUE. We must also find better ways to handle the big data required for precision management. Digital agriculture can already convert N prescription maps to CO_2 emissions abatement, as shown in Figure 4, but scientists, extension services, private consultants, and farmers require stronger collaboration and communication to increase adoption of digital technologies and to tailor incentive programs based on verified N_2O emission reductions.

2. Improve quantitative modeling

Fluxes of N_2O are notoriously difficult to forecast; plant-soil-atmosphere models rarely predict daily emissions from novel sites with any more than 20% accuracy. This is partly due to model structure. For example, we cannot yet include information about microbial diversity, which influences fluxes differentially among sites. Further, we have too few sites with continuous long-term data that can be used to further refine and validate models, whether process-based, AI-powered, or hybrid (e.g., Saha et al., 2021). A set of long-term sites in different locations where N_2O can be measured and with associated agronomic and environmental parameters could provide both the long-term measurements needed to build better models and a place to test new mitigation technologies. Moreover, dynamic process-based models coupled with high-resolution in-season remote sensing imagery would better capture spatial and temporal variation of N plant demand and N supply. This would, in turn, help improve site-specific model input and parameters as well as identify proxies for scaling AI powered algorithms or process-based biogeochemical and crop models.

3. Improved measurement technologies.

N_2O fluxes are usually measured in small chambers placed on the soil surface and sampled at weekly to monthly intervals. The limitations of such systems are well-known, even when automated to allow more continuous measurements (e.g., Grace et al. 2020). Sensors that allow continuous sampling over hectare-size areas are available but unsuitable for sites without line power, which limits their use. Developing low power, open-path eddy covariance sensors similar to those used for ecosystem CO_2 fluxes would avoid the limitations inherent to chamber sampling and greatly improve the accuracy and precision of flux measurements from different systems and management regimes, thereby also improving model development.

Conclusions

N_2O is a potent greenhouse gas accumulating in the atmosphere at increasing rates due to more intense agricultural production. Most emissions are from agricultural lands and the downstream ecosystems that receive excess agricultural nitrogen inputs. Nitrogen availability is the single best predictor of soil N_2O emissions; emissions accelerate when rates of soil nitrogen inputs exceed crop needs. Reducing excess reactive nitrogen in the environment by increasing system-scale NUE is the single best available strategy for abating N_2O emissions. This can be achieved in part by reducing nitrogen fertilizer rates to better match crop nitrogen needs through more aggressive 4R management and process-based crop modeling. A more effective strategy is to use variable rate (precision) geospatial fertilizer technology to apply nitrogen, avoiding subfield areas in up to 27% of farmland in the U.S. Midwest with consistently low yields. Alternatively, these subfield areas could be planted to perennial vegetation like cellulosic bioenergy crops or conservation strips. Better minimizing N_2O fluxes requires research to improve field scale NUE, quantitative modeling including validation and verification sites, and better field measurement technologies.

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FIGURE LEGENDS

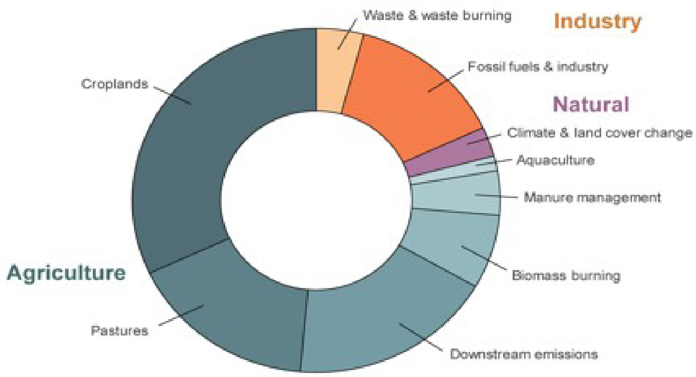

Figure 1. Global sources of anthropogenic N2O. Calculated from Tian et al. (2020).

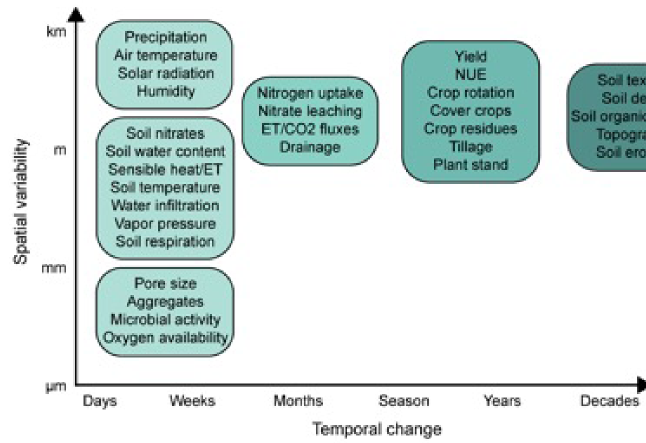

Figure 2. Cropping system factors that control nitrification and denitrification vary across temporal (hours to centuries) and spatial (nm to landscape) scales.

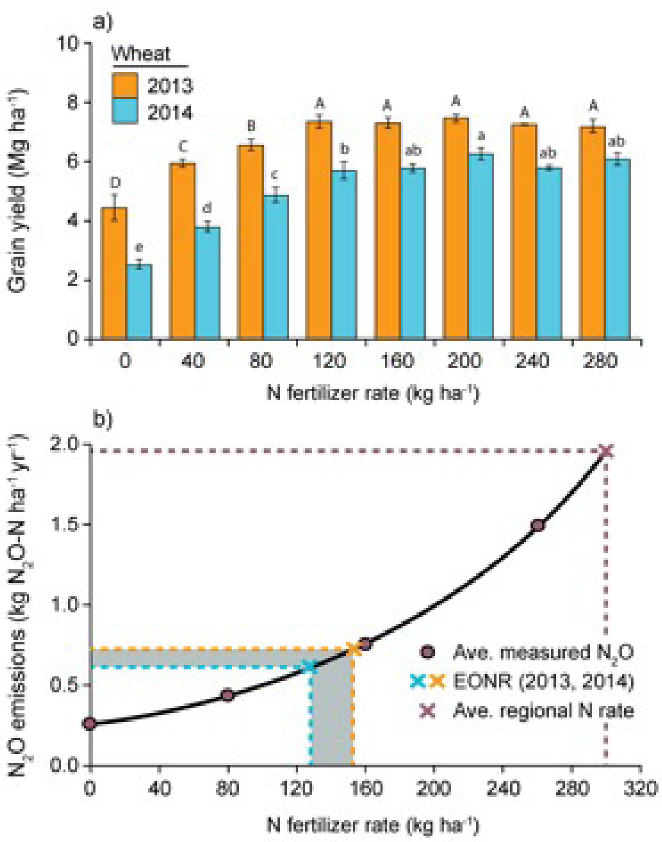

Figure 3. Yield (a) and N2O emissions (b) response to nitrogen fertilizer additions in an irrigated wheat system in the Yaqui Valley, Mexico (Millar et al. 2018). The shaded area in (b) illustrates N2O emissions in 2013 and 2014 if fertilizer had been applied at rates to achieve yields at the optimal economic return to nitrogen rate (EONR; 130-150 kg N ha-1 yr-1). In contrast, the upper dashed line in (b) shows N2O emissions at average rates of nitrogen fertilizer application for the region (300 kg N ha-1 yr-1).

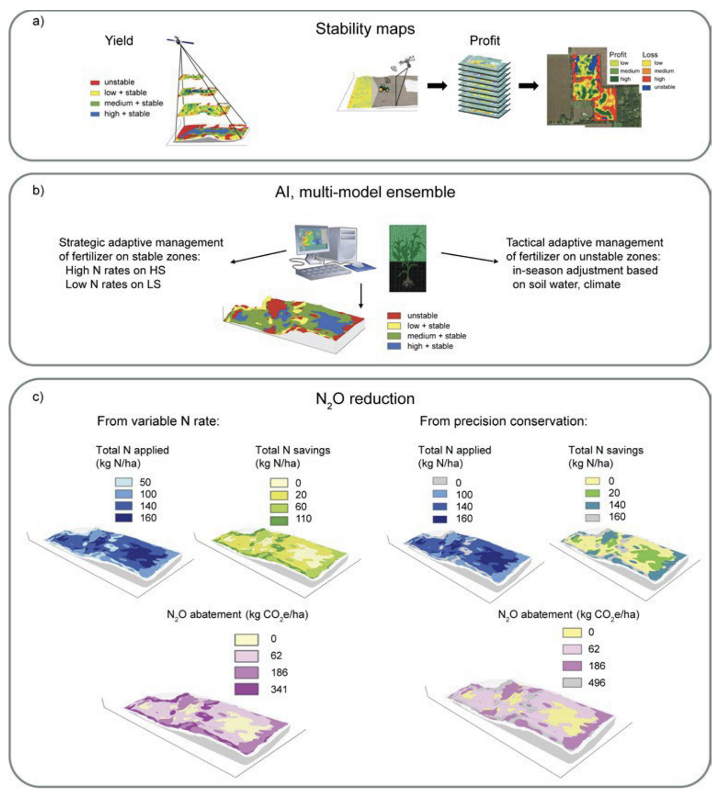

Figure 4. A geospatial systems approach to link (a) stability and profit maps with (b) AI and multi-model ensembles to (c) reduce N2O emissions through variable rate N applications or precision conservation practices.

Chapter 4: The Challenges and Opportunities for Closing the Row Crop Yield Gap

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Linkage to Greenhouse Gas Negative Agriculture

Understanding the path toward a greenhouse gas negative agricultural system requires we consider and evaluate how each component of the production system can contribute. In row crop production the inability to achieve crop yields that are close to the genetic potential represents an opportunity to reduce the greenhouse gas footprint. In this chapter, we explore closing the yield gap as a component of developing greenhouse gas negative agriculture.

Background: Yield Gaps, Yield Potential, and Actual Yield

Yield gaps represent a concept that evaluates crop performance in a biophysical and statistical quantification of historical yields across field, county, state, or national scales. The framework for yield gaps is based on three components: potential yield (Y_p), attainable yield (Y_{at}), and actual yield (Y_a). These components have been defined by different individuals and we can trace it back to Boyer (1982) who defined Y_p as the crop yield when grown in a non-stressed environment. Later, Evans and Fischer (1999) developed a more inclusive definition for Y_p as "the yield of a cultivar when grown in environments to which it is adapted; with nutrients and water not limiting; and with pests, diseases, weeds, lodging, and other stresses effectively controlled". This could be considered as the genetic potential of a crop cultivar or hybrid, and separate Y_p estimates are usually calculated for irrigated *versus* dryland crops. Meanwhile, Y_a is obtained by growers in commercial operations during a growing season, and is inevitably smaller than the Y_p due to sub-optimal management practices or abiotic or biotic stresses that limit production. Yield gap (YG) is then defined as the difference between Y_p and Y_a. Cassman et al. (2003), Lobell et al. (2009), and Liu et al. (2017a) demonstrated that YGs provided a metric to quantify how Y_a varied relative to Y_p and the likely reasons for not being able to achieve Y_p. Fischer et al. (2014) provided a detailed analysis of Y_p and Y_a across a large variety of crops important to the world's food supply. They showed that Y_p and Y_a yields were on a positive trend, and YG were slowly closing.

Yield gap analysis is challenged by the inability to accurately assess Y_p, which has led to defining Y_{at} (i.e., the greatest observed yields at a given level of resource availability). There is the opportunity to utilize county and state level yields available from the NASS databases; however, long-term records of field level yields are often difficult to obtain. The NASS databases at the county and state level are available for many crops with a documented procedure of how these values were obtained which provides the ability to quantify yield gaps over different regions and times. Attainable yield can be estimated through statistical analyses by assuming the highest yield observed was subjected to very few nonlimiting conditions (e.g., using boundary function of yield versus water available, French and Schultz, 1984). It has been suggested that crop models could be used to simulate Y_p (Van Ittersum et al., 2013); however, crop model assumptions and different philosophies regarding "bottom-up" or "top down" approaches to long-term simulations still cause a large degree of uncertainty in Y_p estimates (Grassini et al., 2017). Crop models don't include biotic stresses as part of the yield limiting factors so Y_p is determined by minimizing abiotic stresses. Using county level crop yield data, Egli and Hatfield (2014a; 2014b) used an upper quantile analysis to obtain the upper frontier of observed county level yields for soybean (Glycine max (L.) Merr.) and maize (Zea mays L.) across Iowa, Kentucky, and Nebraska to show the impact of soil quality on Y_{at} and Y_a and the YG (Hatfield et al. (2017) extended this approach to the Midwest using National Agricultural Statistics Service (NASS) data to evaluate the causes of the YG for maize and soybean in the Corn Belt. Meanwhile, in small grains, Hatfield and Dold (2018) showed the primary factor affecting the wheat YG across the Great Plains of the United States was the availability of soil water during the grain-filling stage of growth. These results were similar to those in the same region by Patrignani et al. (2014) using county-level data and seasonal water availability; and those by Lollato et al. (2017) using crop simulation models.

Research on the yield gap has focused on the factors that affect yield, and consideration of these factors relative to greenhouse gas negative systems have not been extensively addressed; however, we can use our understanding of these factors to suggest a path toward a greenhouse gas negative system. To reduce the YG and obtain a greenhouse gas negative agricultural system will require we focus on the interactions among genetics (G), environment (E), and management (M) using the G × E × M concepts outlined by Hatfield and Walthall (2015) and Beres et al. (2020). Fischer et al. (2014) outlined steps for increasing Y_p based mainly on the increased capture of solar radiation coupled with the harvest index; however, these increases would benefit from increased photosynthetic efficiency. If there is an effort to increase Y_p without an effort to increase Y_a, then the YG will increase rather than decrease. On the other hand, systems in which Y_a approaches ~70-80% of the potential yield usually show yield stagnation and further yield improvements are not economical (Lobell et al., 2009). Thus, increasing Y_a while moving toward greenhouse gas negative agriculture will require a focus on enhancing the genetic capability of capturing more carbon within the growing season and among growing seasons to improve Y_p, coupled with management of agronomic factors that would increase productivity, yield stability, and reducing greenhouse gas emissions. There are numerous pathways to enhancing both Y_p and Y_a that will affect the yield gap; however, these will require a transdisciplinary effort across disciplines embracing both genetics and management as an integrated solution.

Genetic Approaches To Increase Yield Potential

To increase Y_p and reduce the YG, research must identify and utilize genetic variation and test transgenic approaches for improving traits that determine Y_p and enhance resilience to abiotic and biotic stresses (Xu et al., 2017). In subsequent sections, the role of management practices on closing the YG will be discussed. *Promising genetic approaches then need to be tested under management practices consistent with greenhouse gas negative agriculture to test their potential for contributing to reductions in greenhouse gas emissions*. One potential strategy to increase Y_p is to improve photosynthesis. There has been a resurgence of research to improve the photosynthetic efficiency of crops over the past 10-15 years, justified by the observation that the efficiency of converting light energy to photosynthesis is far below the theoretical maximum. Monteith (1977) proposed the following:

$$\text{Total biomass} = \sum_{\text{planting}}^{\text{maturity}} QIE$$

Where *Q* is the solar radiation incident on the canopy from planting to maturity, *I* is the interception of solar radiation by the crop canopy and *E* is the total dry matter content produced per unit radiation interception. The equation has been extended to express theoretical yield potential (Y_p) as a function of the efficiency of radiation interception, the efficiency of conversion of light energy to biomass, and the efficiency of partitioning of harvest index (Zhu et al., 2010). In this equation, the efficiency of CO_2 uptake is contained in the conversion factor to biomass. This leaves improving the efficiency of carbon assimilation as the component of Y_p with the greatest possibility for improvement (Zhu et al., 2010). Efforts identifying genetic variation in photosynthesis could be used to improve crop performance (Flood et al. 2011; van Bezouw et al., 2019; Faralli & Lawson 2020). Additionally, specific transgenic modifications to photosynthesis are identified by increasingly sophisticated mathematical models of photosynthesis that identify control points for improvement and estimate Y_p and efficiency gains from specific changes to photosynthetic enzymes and processes (Zhu et al., 2007, 2008, 2010; Long et al., 2015; Yin & Struik, 2017; Zhao et al., 2020). There are theoretical ways to increase the efficiency of photosynthesis, from enhancing light capture to increasing the concentrations and efficiencies of enzymes that assimilate CO_2 and regenerate substrates for the Calvin-Benson cycle to enhancing CO_2 diffusion from the atmosphere to the chloroplast (Bailey-Serres et al., 2019; Ort et al., 2015; Zhu et al., 2010). Some strategies to improve photosynthetic efficiency have the added theoretical benefit of improving nitrogen and water use efficiency (Long et al., 2015), which have greater potential to contribute to greennhouse gas negative agriculture if managed appropriately. Moving toward carbon negative agriculture requires a critical assessment of the practices that increase the efficiency of external inputs and the capability of increasing the crop's carbon assimilation.

Light absorption by chlorophyll molecules is the first step of photosynthesis, and theoretically it is possible to expand the wavelengths that higher plants use to capture light energy by substituting photosystems from bacterial systems that contain different chlorophyll molecules (Ort et al., 2015). Light capture and photosynthesis can be improved by accelerating the rate that leaves move from a photoprotected state in high light to a non-photoprotected state in low light (Murchie & Niyogi, 2011). Under high light, plants increase their capacity to dissipate excess energy as heat (non-photochemical quenching, NPQ) to protect photosynthetic membranes from damage caused by singlet oxygen. However, slow relaxation of NPQ when light levels are lower than saturating causes unnecessary dissipation of light energy that could be used for photosynthesis (Zhu et al., 2004; Murchie & Niyogi, 2011). A breakthrough in manipulating NPQ and increasing photosynthesis was recently made by over-expressing xanthophyll cycle enzymes (vioxanthin de-epoxidase (VDE) and zeaxanthin epoxidase (ZEP)) and photosystem II subunit S (PsbS) in tobacco. Transgenic tobacco plants with increased VDE, ZEP and PsbS increased photosynthetic carbon assimilation under fluctuating light conditions and improved above-ground biomass by 15% in field experiments (Kromdijk et al,. 2016). This proof-of-concept in tobacco shows potential for improving Y_p in other row crops.

Much of the inefficiency of photosynthesis is attributed to ribulose-1,5-bisphosphate carboxylase/oxygenase (Rubisco), the notoriously slow, bifunctional enzyme that fixes atmospheric CO_2 (Whitney et al., 2011). There are ongoing efforts to identify more favorable Rubisco enzymes with higher catalytic rates or greater specificity for CO_2 in nature and in crop germplasm (Galmes et al., 2014; Hermida-Carrera et al., 2016) that could be used for improving Rubisco in crops. Increasing the expression and content of Rubisco and its assembly factors have been successful in improving maize photosynthesis and yield (Salesse-Smith et al., 2018), and increased Rubisco small subunit expression increased rice yield and nitrogen use efficiency under high nitrogen fertilization (Yoon et al., 2020). The dual function of Rubisco as an oxygenase and subsequent photorespiration results in a significant decrease in efficiency of C_3 plants, estimated to suppress soybean yields by up to 36% (Walker et al., 2016). Insertion of enzymes to metabolize glycolate (the toxic byproduct of the Rubisco oxygenase reaction) in the chloroplast increased tobacco biomass by up to 40% under field conditions (South et al., 2019). Other transgenic approaches to increase the regeneration of Calvin Cycle intermediates through over-expression of have shown promise in tobacco and wheat (Lefebvre et al., 2005; Driever et al., 2016).

The potential for any photosynthetic manipulation to increase Y_p requires moving the genes/transgenes to high-yielding germplasm and testing the performance of improved or transgenic lines in different environments (Sinclair et al., 2019). Thus, the research to improve photosynthetic efficiency to date shows proof-of-concept and promise, but not yet commercial success in crops. If realized, improved photosynthetic efficiency could increase Y_p, and could help close the YG if improvements in photosynthetic efficiency also result in greater productivity under stress. Currently, YG in Midwestern row crops like corn and soybeans are caused by temperature or precipitation stress (Hatfield et al., 2017). Improved photosynthesis and productivity under high temperatures and drought stress are therefore also needed to close the YG (Ainsworth & Ort, 2010; Leakey et al., 2019), especially as global climate change will exacerbate temperature stress and precipitation extremes, and widen the YG (Hatfield et al., 2017). Additionally, the potential for specific changes to photosynthesis to have trade-offs at high temperatures or under water deficit conditions need to be studied.

There is genetic variation in the temperature response of photosynthesis within crop species (Sharma et al., 2012) and greater mechanistic understanding of processes controlling temperature response has identified promising targets to improve photosynthesis at high temperatures (Perdomo et al., 2017; Slattery & Ort, 2019; Degen et al., 2020). Rubisco activase is a key target, as it is sensitive to moderate increases in temperature and limits the proportion of active Rubisco at higher temperatures (Crafts-Brandner & Salvucci, 2000). Introducing a thermotolerant Rubisco activase from a wild relative of rice into domesticated rice improved growth and yield at higher temperatures (Scafaro et al., 2018). Photorespiration increases at higher temperatures because of lower specificity of Rubisco for CO_2 vs. O_2 and proportionally greater decreases in the solubility of CO_2 compared to O_2. Therefore, Rubisco is a key target for improving thermotolerance (Prins et al., 2016) and manipulations that decrease photorespiration theoretically have greater benefits at high temperatures (Walker et al., 2016; Slattery & Ort, 2019). The regeneration of RuBP will also become increasingly limiting to the rate of photosynthesis with higher atmospheric CO_2 concentrations and higher temperatures. Targeted over-expression of enzymes that regenerated RuBP has potential for improving photosynthesis and soybeans expressing a cyanobacterial fructose-1,6-bisphosphatase/sedoheptulose-1,7-bisphosphase showed increased photosynthesis at elevated CO_2 and elevated temperature conditions (Kohler et al., 2017). While the potential benefits and tradeoffs of modified physiological traits across several locations and years can be initially assessed by modifying such processes in mechanistic crop simulation models (e.g., Sinclair et al., 2010; Messina et al., 2015; Sciarresi et al., 2019), field-testing of strategies to increase thermo-tolerance of photosynthesis are needed to ensure that trade-offs that benefit productivity under one set of environmental conditions don't result in lower productivity under others. Critically, future experiments need to explicitly design field tests of the potential photosynthetic strategies to contribute to carbon negative agriculture.

Management and the Yield Gap

Genetic improvement continues to increase the Y_p of most cultivated crops, e.g., corn, soybean, rice, wheat (Fischer et al., 2014); however, attaining this Y_p requires improved management, e.g,, planting dates, seeding rates, fertilizer management (Beres et al., 2020). For crops grown in regions characterized by large YG, environment, and management can account for a much larger proportion of yield variability than genetics. For instance, management including environment accounted for 44-77% of yield variability of winter wheat in the US. Great Plains (a crop system with large YG; Lollato et al., 2017; 2019), whereas genetics accounted for 1-8% (Munaro et al., 2020). While the Munaro et al. (2020) results only evaluated well-adapted, commercially available cultivars – perhaps reducing the genotypic effect when compared to most studies calculating the genetic yield gain of modern cultivars against historical ones (e.g., Bell et al., 1995; Nalley et al., 2008; Maeoka et al., 2020; Lollato et al., 2020) – they align well with results using similar protocols for other growing regions (Cullis et al., 2000; Friesen et al., 2016; Mohammadi et al., 2010) and highlight the importance of prioritization of research and development (R&D) investments in improved agronomic management to reduce the YG.

A number of management practices and their interactions (e.g., seeding date and rate, in-season nutrient and pest management, etc.) can narrow the YG, and is beyond the scope of this work to extensively review them. Alternatively, we will highlight a few management practices that show greater potential for increases in both Y_p and Y_a, as well as research methods that allow for exploration of several management practices at a time. Alley and Roygard (2001) suggested that practices could be divided into either "yield building" or "yield protecting" factors. Using an analog concept but within the context of YG analysis, we propose that some management practices affect the Y_p of the crop while others will determine the Y_a within the constraints imposed by the previously established Y_p.

Management practices affecting yield potential

Beyond selecting an adapted crop species and cultivar (Evans and Fischer, 1999), management practices that impact the environment in which the crop is to be established, or the availability of soil water in rainfed environments, affect the crop's Y_p. The example of sowing date on Y_p– calculated through the boundary function approach – is perhaps the clearest one. For soybeans grown in North-Central U.S., delays in sowing date cause a linear decrease in yield potential in as much as 33 kg ha^{-1} day^{-1} (Grassini et al., 2015; Rattalino Edreira et al., 2017) owing to decreases in the duration of the crop cycle and of the critical period (i.e., the most important period for a crop's yield determination, which in soybeans include the interval from R3 to R7; Rattalino Edreira et al., 2020). Meanwhile, the Y_p of winter wheat in the U.S. Great Plains typically shows a convex-quadratic response to sowing date (Munaro et al., 2020), with yield reductions ranging from nil to 314 kg ha^{-1} day^{-1} depending on region and direction (i.e., advancements or delays). The penalties to the Y_p result from a number of abiotic factors (Sacks et al., 2010), perhaps the most important being the environment experienced by the crop: Early sown winter wheat allow for a fall environment inductive to excessive growth and unproductive water and N consumption (van Herwaarden et al., 1998); whereas late-sown crops have decreased fall tillering potential (Dahlke et al., 1993) and root growth (Hammon et al., 1999), shortening the duration of the critical period (for wheat, the critical period being that between beginning of stem elongation and beginning of grain fill; Fischer, 1985) (Cossani and Sadras; 2021). These conclusions were reached through on-farm surveys or existing datasets and did not utilize new replicated field experiments suggesting that existing datasets could be effectively utilized to establish relationships and chart a path forward for new research efforts.

Crop rotation can impact a crop's Y_p. In some cases, this impact can relate to the modulation of the environment experienced during important phenological stages of crop development. For example, when winter wheat is sown immediately following the harvest of soybeans, yield potential is reduced owing to a delayed sowing time, worsening the environmental conditions to which the crop is exposed (e.g., warmer and shorter critical period; Staggenborg et al., 2003). This yield reduction is also partially explained by a reduced soil water profile available at sowing compared to other rotations where the crop follows a summer fallow (Patrignani et al., 2012; Lollato et al., 2016). On the other hand, the presence of alternative crops in the rotation to disrupt continuous cropping (i.e., "break crops") can improve Y_p in part due to less water use by the break crop (Cutforth et al., 2013; Larney and Lindwall, 1994).

Crop rotations can affect Y_p due to more complex and longer-term processes. For example, yield of corn and grain sorghum were greater when following winter wheat in the semi-arid High Plains of the U.S. due to greater available soil water at sowing compared to other previous crops (Schlegel et al., 2019), reflecting improved soil physical properties related to efficient water use (i.e., infiltration rate and water-stable aggregates; Stone and Schlegel, 2010) and greater carbon storage (Doyle et al., 2004) when wheat was included in the rotation. Similar benefits resulting from the inclusion of wheat in the rotation were shown by expanding corn/soybean rotations near the U.S. Corn Belt (Janovicek et al., 2021). Other reasons behind the effects of expanded rotations on the crop's Y_p include improvements in soil health (Al-Kaisi et al., 2015) and beneficial rhizosphere microorganism communities (Turco et al., 1990; Rosenzweig et al., 2018a, b), as well as suppression of pathogenic soil microorganisms that could be a sink for crop photosynthates (Harris et al., 2002; Schillinger et al., 2018). These factors will become the foundation for carbon negative agriculture because each of these factors increase the ability to store carbon in the soil and remove it from the atmosphere. Increasing both yield potential and closing the yield gap will increase the amount of carbon extracted from the air and captured in the roots and shoots; however, this increased capture must be coupled with management practices that reduce the loss of carbon from the system.

Management practices to improve actual yields within a given potential yield

Beyond its effects on the Y_p of the crop, expanded and intensified crop rotations provide ecosystem services that can improve Y_a compared to monocultures. For instance, winter wheat grown succeeding a winter canola crop in the U.S. central Great Plains yielded ~12% more than when following another winter wheat crop (Bushong et al., 2012), with yield gains of similar magnitude reported in other regions of the U.S. and the world (Smiley et al., 1994; Seymour et al., 2012; Kirkegaard et al., 2008; Kirkegaard and Ryan, 2014) and for break crops other than canola (Arshad et al., 2002; Krupinsky et al., 2006; Miller et al., 2003; Williams et al., 2014). Candidate reasons behind this improvement in actual yield include better control of troublesome grass weed species (Bushong et al., 2012) and soilborne pathogens (Smith et al., 2004; Angus et al., 2015) that are common in cereal monoculture systems.

The benefits of legumes breaking a continuous cereal monoculture system were also shown for maize in the U.S. Corn Belt, which yielded more when following soybean as compared to monoculture maize (Crookstone et al., 1992; Meese et al., 1991; Farmaha et al., 2016; Grassini et al., 2011; Stanger et al., 2008; Sindelar et al., 2015). From a

Y_a perspective, these benefits were associated with pest control and difficulties in achieving a good stand establishment for maize in monoculture (Bullock et al., 1992; Farmaha et al., 2016). Finally, intensified cropping systems provide other ecological processes that can help replace external chemical inputs in a system where fallow periods are part of the rotation (Rosenzweig et al., 2018a, b). For instance, crop rotations, including legumes as cover crops, can increase the recovery of the applied N fertilizer, ultimately affecting actual yield or reducing the need for N input (Gardner and Drinkwater, 2009), as discussed in the nutrient section below.

Most of the other management decisions a producer makes during the growing season (i.e., seeding rate, nutrient and pest management, etc.) will ultimately affect the Y_a of the crop, and numerous research papers investigated their individual effects for each crop. Our objective was not to thoroughly review the impact of all possible management practices on reducing crop YG; instead, for the remaining practices and their effects on yield, we will offer insights about methodological aspects for future R&D investments into agronomic management. There are challenges and opportunities for greenhouse gas mitigation in cropping systems as described by Smith et al. (2008) and adapt to climate change (Hatfield et al., 2011) while enhancing crop productivity.

First, we note that a systems approach in which individual practices tackle different yield-limiting factors (e.g., see Box 1 in Sadras et al., 2020) is needed for transformative changes (Vermeulen et al., 2018). Yet, field experiments usually evaluate the effect of one or a few management practices at a time due to limits regarding the size and complexity of the experiment, as well as costs associated with establishing field experiments in a representative number of environments (Van Ittersum et al., 2013). As a result, recent studies attempted to quantify the benefits of an intensive management "package" where several practices are combined and applied in a prophylactic manner to avoid yield-limiting factors in replicated field trials (e.g., for soybeans, Orlowski et al., 2016 and Marburger et al., 2016; for maize, Ruffo et al., 2015 and Balboa et al., 2019; and for wheat, Jaenisch et al., 2019, De Oliveira Silva et al., 2020, and Roth et al., 2020). While these studies provided excellent empirical data for the biophysical limits of the crops evaluated (i.e., the Y_p under the most intensive management), they usually failed to improve current recommendations because of the complexity of the interactions between management and environment, often concluding that an integrated pest management based on crop scouting should be pursued.

An alternative to costly replicated field experiments that has gained momentum in recent years is the use of surveys of management practices adopted in a large number of commercial fields and their respective grain yield (e.g., Grassini et al., 2015; Rattalino Edreira et al., 2017; Mourtzinis et al. 2018; Lollato et al., 2019). While this approach does not establish cause and effect relationships among management practices and yield due to the unstructured nature of the dataset, it offers several advantages such as (i) assessing current on-farm management relative to recommended practices, (ii) discerning the yield-limiting factors and the relative contribution of individual management and production factors at the level of commercial-scale fields (in contrast to small experimental plots), (iii) documenting the range of cost-effective management practices being used by high-yield and low-yield producers, and (iv) quantifying field- and regional- resource use efficiency. Information from on-farm surveys could be used to help develop representative (i.e., reflecting the management currently adopted in commercial fields of different management intensity levels) and relevant (i.e., how can technology currently being adopted in the fields with the smallest YG benefit the remaining fields) treatments to be tested in replicated experiments to answer more specific questions. As data from field experiments becomes available, there is the potential for increasing the use of various data mining techniques to quantify how changes in management can affect both closure of the yield gap and resilience to climate change.

Finally, we note that the availability of sensors and mapping and tracking technologies adopted in commercial fields resulted in the so called "big data", infinitely increasing the opportunities to address several agricultural issues, including those related to management practices to reduce the YG (Coble et al., 2018). The opportunity to collect and explore large datasets for insights into opportunities to narrow the YG has never been greater, including those to increase within-field yield variability by investing more inputs where the Y_p is greater (e.g., Schwalbert et al., 2019). However, as suggested by Sadras et al. (2020), there is a need to collect and explore these datasets using a hypothesis-driven approach; otherwise, these datasets might follow the examples of functional genomics and fail to deliver improvements in crop production at the field level – at least to date (Porter et al., 2018).

Nutrient management to enhance potential and actual crop yield

A review of greenhouse gas emissions concluded that ecological intensification of agricultural production could close the YG, restore soil carbon, and lower greenhouse gas losses per unit of yield through judicious use of the 4Rs and appropriate location specific management (Snyder et al., 2009). Fixen (2020) described the background of the 4Rs' for nutrient management (right rate, right form, right placement, right timing) and its potential for agricultural application. It takes optimal decision-making in all 4Rs, the development of new products that reduce greenhouse gases and synchronize nutrient release with plant uptake, the advancement and adoption of new technologies and equipment, fair and equitable economic incentives, and the entire agricultural enterprise using these tools, technologies, and products to reach a greenhouse gas negative future with optimized yield goals. Increases in soil C also offer an opportunity for greenhouse gas negative. Most of the research evaluating increases in soil C has focused on no-tillage practices and the associated increase in soil surface residue (Powlson et al., 2014; Giller et al., 2015). However, recent evidence suggests that an appropriate balance of nutrients such as N, P, and S (Kirby et al., 2011; Richardson et al., 2014) either through fertilizers and/or through the use of N2-fixing legumes and pastures (Da Silva et a., 2014) are crucial for long-term increases in soil C (Giller et al., 2015). Farmer use of nutrients (4Rs) is often informed by habit (Osmond et al., 2013), and adoption of best nutrient management practices has many determinants (e.g., area farmed, age, capital, education, experience, income, networks, land tenure awareness; Prokopy et al., 2008). There is often a divergence between nutrient management plans and implementation due to many factors ranging from weather conditions to time constraints (Cabot and Nowak, 2005; Osmond et al., 2014; Tao et al., 2014). The literature on fertilizer or nutrient decision making shows a complex assemblage of farmer behavior related to nutrient application. Producers, farm groups, commodity producers and distributors, the business community, and state and local agencies must be involved in delivering advances in nutrient management that are cost, time, and environmentally effective.

Throughout the past 40+ years, maize yields have increased by 1.8 bu acre-1 yr^{-1} (25 kg ha^{-1} yr^{-1}); attributed to soil testing and associated fertilizer and lime applications, crop breeding, conservation tillage, and integrated pest management; however, a YG remains (Cassman et al., 2006). Closing any YG that arises due to issues associated with nutrient availability will require increased nutrient use efficiency by synchronizing nutrient supply with crop demand as affected by weather and field conditions through detailed 4R management as the application of the right nutrient rates, sources, placement, and timing. Since nutrient management decisions (the 4Rs) are rarely independent of each other and affected by other agronomic management factors, as well as economics, their use as a system is necessary to reduce biophysical yield limitations while lowering the N and C footprints (Wang et al., 2020). Closing the YG, especially in the context of becoming C neutral, will require more ingenuity relative to our current nutrient recommendations, greater precision of rates and timing, new nutrient formulations and products (i.e., “smart fertilizers”), better digital tools, analytics, and models to predict yield and characterize the environment, and advanced precision equipment. One of the hardest factors in this quest, but one that could provide immediate results and outcomes in the future, is continuing and expanding efforts to ensure farmers can operationally and economically implement 4Rs now and into the future. Utilization of efficient nutrient management practices can contribute to carbon negative agriculture by increasing the potential of the crop to capture more carbon and when linked with management practices improve the efficiency of nutrient use and reduce the loss of carbon from the soil or nitrous oxide (Chapter 3) then the path toward carbon negative agriculture can become a reality.

Rate

Data from The Fertilizer Institute (2020) suggests that soil test results for P or K show fertilizers are needed 50% or more of fields found in 14 to 16 states, respectively, while many states have sufficient to excess levels. Producers must ensure adequate but not over-fertilization to optimize production, especially as P and K resources become more limited worldwide. Additionally, to close the YG, we must ensure soil test recommendations provide up-to-date and scientifically valid rates, which are consistent across state lines and represent the yield potential of new varieties (Hergert et al., 2015). Soil testing correlation and calibration research has been underfunded for years throughout most of the U.S. and has made it impossible to determine whether recommendations are sufficient to close the YG (Voss, 1998). New initiatives, such as the Fertilizer Recommendation Support Tool (FRST), provide a framework for updating and harmonizing soil test recommendations (Lyons et al., 2020) to optimize yield while minimizing environmental impacts. Publicly funded, nationally linked soil testing decision tools are critical to progress toward real-time, in-field nutrient status identification.

Although N rates can be determined in some western states, e.g., Nebraska, through soil testing (Schepers et al., 1986), most N rate decisions are independent of laboratory analysis and based on expected yield. However, yield and N rates are seasonally affected by weather, management (e.g., population and variety), and soils. A recent meta-data analysis demonstrates that the selection of an N rate to optimize yield is a location by year determination (Dhital and Raun, 2016), which is difficult for producer N-rate decision making given the dependence on post-season analysis. Many studies have shown increased nitrous oxide emissions with higher N application rates. Matching N needs to ensure maximum yield while minimizing excess N is critical but difficult as N rate is a moving target (Shcherbak et al., 2014; Banger et al., 2020). Models to predict N rates have been developed by private and public sectors; some years these models predict N rates better than current recommendations, sometimes they do not, but in general they reduce uncertainty and risk (Morris et al., 2020; Tremblay et al., 2012). Significant research on in-field sensor technologies has allowed farmers to estimate crop N status in real-time and adjust rates as-needed if farmers have access for late-season N adjustments (i.e., technology, high clearance equipment, aerial application, or supplemental irrigation), and if the yield benefit outweighs the increased time management and costs (Osmond et al., 2014; Morris et al., 2020). Model improvement and imaging, especially when multiple spatial and non-spatial data sources are integrated, should significantly increase precision in N rates and aid in understanding and management of yield stability (Basso et al., 2011; Franzen et al., 2016; Antle et al., 2017). It is expected that over the next 30 years, as the tools and technologies develop and become increasingly available, they will greatly enhance N rate (and timing) decision-making to close the YG and reduce N losses.

Timing

Timing is linked to rate, placement, and source, especially for N, as most other nutrients are generally applied pre-plant or at planting. Models and sensor technologies both can and will aid in fine-tuning timing decisions. The development of enhanced sensors capable of covering different soils in the field and transmitting the results to a central location offers the potential for improving nutrient management. Ensuring that fertilizer applications and forms are matched more closely to crop needs and understanding how changing the carbon status of the soil remains a challenge; however, offers the potential of being able to positively impact productivity and environmental quality.

Form

Nutrient form, particularly for N, can affect greenhouse gas production, nutrient efficiency, and yield. Often, farmers may have little choice in the fertilizers available to them or may be constrained by the type of equipment accessible (Personal comm. O'Connell). Unfortunately, timing may be pre-determined if farmers have limited access to different fertilizer forms (e.g., anhydrous ammonia vs urea ammonium nitrate, UAN) or equipment. In the future, appropriate forms of fertilizers and the equipment necessary to deliver the fertilizer will be critical to utilize models and sensor decision-making around rate, placement, and timing. Enhanced efficiency fertilizers (EEFs) have had historical use for several decades. Although urease inhibitors and nitrification inhibitors may reduce atmospheric N losses depending on the fertilizer type, placement, and soil and crop conditions (Thapa, et al., 2016; Woodley et al., 2020), data typically show inconsistent yield increases (Mitchell and Osmond, 2012; Thapa et al., 2016; Woodley et al., 2020). There is significant interest in new EEF product formulations or smart fertilizers as evidenced by the Next Gen Fertilizer Challenges competition sponsored jointly by USEPA and USDA (2020), as well as novel fertilizers (Lui et al., 2017; Ranieri et al., 2021) and products, including biologics, to reduce environmental impacts, such as P fertilizers that may release phosphorus more slowly or increase the availability and solubility (Fertahi et al., 2019). The expectation of these products is to close the YG while minimizing environmental losses by synchronizing nutrient release with crop uptake and soil water availability, thus increasing nutrient use efficiency.

Placement

Increased nitrous oxide (N_2O) losses from broadcast N fertilizer have been repeatedly demonstrated (Banger et al., 2020; Woodley et al., 2020), but these are normally surface applied with conservation tilled crops or hay/pasture. Additionally, there is often a relationship between fertilizer form and placement (e.g., anhydrous ammonia is injected into the soil while urea is often broadcast). Form and placement affect nitrous oxide emissions and yield but are mediated by soil systems and crops (Nash et al., 2012; Halvorson and Del Grosso, 2013). Future applications of nutrients should focus on placing nutrients into the soil regardless of agricultural system and/or nutrient source in the most cost-effective method.

Nutrient management (manures and organics to increase soil quality)

Long-term trials at the Morrow Plots (University of Illinois) and Rothamsted Experiment Station (England), both over 100 years old, demonstrate that soil organic carbon is greater in fertilized than in nonfertilized plots. Manured plots, however, protected or built C more than fertilized plots, suggesting the need for animal manures to increase soil organic matter (Weil and Brady, 2017). Animal manures, if managed well, provide multiple benefits, including macro- and micro-nutrients, additional C, and increased microbial activity (Ozlu et al., 2019; Miner et al., 2020). Because nutrients in animal manures are not balanced similar to crop needs, and because animals are frequently raised in relatively small areas that require grain inputs from outside areas (e.g., poultry in the Delmarva Peninsula or swine in southeastern North Carolina), overapplication of some nutrients, such as phosphorus and zinc, often occur and manure shed management is being promoted to balance nutrients (Spiegal et al., 2020). Reducing greenhouse gases from animal manures can be accomplished by covering lagoons to capture methane for energy co-generation, or litter can be burned to generate energy, and the ash used to manufacture fertilizers (Cantrell et al., 2008; Key and Sneeringer, 2011). Phosphorus can be recovered from waste streams, reconstituted into fertilizers, and shipped to P deficient areas (Cantrell et al., 2008; Karunanithi et al., 2015). Liquid systems should be injected into soil to reduce losses of greenhouse gases and rotated so that phosphorus levels do not increase. In the long term, adjustments in the animal-to-land ratio will be required to distribute C widely to balance nutrients; ultimately, manure will need to be treated as a commodity rather than a waste. The animal-to-land ratio represents the amount of manure produced per animal relative to the application rate and requires that we begin to consider more cost-effective methods of manure handling and distribution across the landscape to reduce the carbon footprint of manure utilization.

Improving water use efficiency of crops and the yield gap

Soil water availability is one of the main limitations to crop yield and adequate water during the growing season has been linked to closing the YG (Fischer et al., 2014). Observations in irrigated fields show the YG is quite small (Grassini et al., 2011; Liu et al., 2017b), resulting in a high water use efficiency (WUE) for these systems. In row crops, WUE decreases when soil water is limiting during the grain filling stage, thus increasing the carbon footprint of these crops because of the limitations in the efficiency of grain filling (Hatfield, 2012). Providing water to meet crop demand during grain-filling is critical to high yields and observations of yield monitor data collected from production fields in the Midwest show grain yields are close to Y_p in areas of the field with high soil water availability (Hatfield, unpublished data). These areas of the field have higher soil organic matter and enhanced water dynamics (infiltration, storage, and access by crops) showed this is a path toward carbon negative systems. To achieve these changes in the soil requires producers adopt soil management practices that reduce carbon loss from the soil and enhance soil health through improved soil water storage and infiltration.

Recommendations for Future R&D Prioritization

Relevant management practices impacting Y_p and Y_a of different crops reveals that the often overlooked "management" piece of the G × E × M puzzle can be responsible for a much larger proportion of the yield variability than genotype in cropping systems with large YG. Future R&D investments should thus prioritize agronomic management, both from a standpoint of improving Y_p and Y_a, to ensure YG are minimized to levels that maximize profitability. However, this is not to suggest that we ignore the role that improved genetic and physiological research could have on enhancing Y_p and Y_a and the efforts on enhancing photosynthetic efficiency and resilience to environmental stresses should remain as a high priority – especially in cropping systems where Y_a is already near the Y_p and future yield increases rely on improved Y_p. In the G x E x M concept, we have the least control over the E (soils or weather) factor; however, utilizing genetic material with greater resilience to variation in E under different M practices related to greenhouse gas negative systems could offer insights into these complex interactions. This will require that we begin to incorporate the impacts of pests, e.g., weeds, diseases, insects as yield-limiting factors and the associated carbon dynamics linked with these control measures. This will be further exacerbated by a changing climate directly affecting the crop and indirectly affecting the pest dynamics.

We suggest that prioritization of R&D investments be placed into the evaluation and deployment of expanded and more intense crop rotations due to their effects on improved soil physical and chemical properties that can increase potential and actual yields, as well as provide ecosystem services that potentially reduce chemical inputs, including the use of biological systems that fix nitrogen. These rotations should include new crop species and legume cover crops (i.e., coupling carbon and nitrogen cycles) as these result in the largest benefits to the system (Gardner and Drinkwater, 2009). Caution is needed in dryland environments where cover crops can be detrimental because of the use of limited soil water (Holman et al., 2018). Traditional hypothesis-driven replicated experiments (e.g., Doyle et al., 2004) provide the most impact of elucidating mechanisms of some of the complex interactions; however, sampling of a large number of commercial fields (e.g., Rosenzweig et al., 2018a, b) coupled with data mining and statistical analysis can provide insights into the dynamics of producer decision making to implement different practices.

While replicated field experiments can offer causality and should still be pursued to quantify the effect of management practices on increased actual yields whenever possible, these efforts should be complemented by research opportunities that offer insights from a greater number of explanatory variables (e.g., on-farm surveys) or preclude the need for new and costly research trials, either through the re-utilization of currently existing data (e.g., Munaro et al., 2020), through literature synthesis and meta-analyses (e.g., Gardner and Drinkwater, 2009), or by using satellite data (Lobell, 2013) and crop models (Rosenzweig et al., 2012). Opportunities for the development of site-specific information for narrowing the YG have never been greater owing to big data; however, caution is suggested when developing research questions to collect and explore such data to ensure yield gains are translated to increases in grain yield at the field level and avoid failures-to-deliver as experienced in other disciplines.

Finally, we encourage the development of interdisciplinary and transdisciplinary research groups focused on improved agronomy of different cropping systems. One example is the Wheat Initiative Agronomy Expert Working Group, where researchers around the globe who share a focus on wheat agronomy within their respective programs gathered to develop a global inventory of current wheat agronomy research (Beres et al., 2020). This inventory was used not only to identify synergies among different programs but most importantly, to produce a uniform research priority agenda and to set immediate goals for the working group. The development of similar groups focused on other crops and cropping systems can systematically review the existing efforts to identify gaps and prioritize R&D within each discipline.

Farming systems with improved carbon balance and enhanced Y_p and Y_a, provide a pathway toward closing the YG while increasing soil carbon and restoring soil functionality. Decreasing the carbon footprint and moving toward greenhouse gas negativity of row crop systems while reducing the YG presents a challenge of integrating enhanced physiological efficiencies incorporated into genetics while coupled with management practices that optimize genetic performance. One of the major challenges is the variation in weather across years within the growing season which is one of the primary factors affecting the YG requiring that we increase the capability of all soils to supply water and nutrients as a foundation of resilience in agricultural systems to enhance the effectiveness of management practices. These are not unsurmountable challenges; however, they require that we begin to utilize transdisciplinary teams to develop solutions for this complex if not wicked problem.

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Chapter 5: The Challenges and Opportunities for Animal Protein Production

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Introduction

The global population may reach 9.7 billion by 2050 and 10.9 billion by 2100 (UN Department of Economic and Social Affairs, 2019). About 200 million tons of meat will be needed by 2050 to reach a total of 470 million tons to satisfy food demand (UN FAO, 2009). The estimated increase in global animal protein demand by 2050 and beyond will further strain our food systems, the natural resources that support them, and the people that make them possible. In the face of an already changing climate, we must intensify our focus on innovation in GHG-negative systems. Sharp reduction of emissions of short-lived climate pollutants, especially methane (CH_4), to the atmosphere results in a negative climate state because more carbon will be removed on top of the natural breakdown of those gases.

Animal agriculture production systems provide food, fiber, and other products to consumers and can protect and restore carbon (C) in pasture and rangelands while providing wildlife habitat and maintaining other ecosystems (Sanderson et al., 2020). Animal agriculture contributes an estimated 3.9% of the 5.5 Gt CO_2-eq GHG emissions from the U.S. (USEPA, 2024). Livestock systems, however, are complex and important to world food systems, so pathways toward GHG-negative livestock supply chains must consider system interactions and potential unintended consequences. GHG negative animal agriculture production should not sacrifice and ideally improve land, air, and water quality, water use, food security, animal health and well-being, worker safety and satisfaction, public health, and value chain profitability. To achieve a sustainable outcome, in addition to producing more meat, milk, eggs, and fiber, unintended consequences must be avoided on the journey to climate negative production systems. In the following narrative, we will consider how we can achieve GHG negative animal agriculture production, although many of these suggestions require additional analysis through a systems lens to ensure the overall sustainability of the system.

We focus on technologies and practices for reaching GHG negative emissions in beef (meat and dairy), from the field to farm gate, where most GHG emissions from animal systems occur. Other animal production systems certainly contribute GHG emissions but beef is the dominant contributor and thus the appropriate focus for moving to GHG negative agriculture. Pork and poultry GHG emission reduction strategies are also addressed. We define negative climate emissions as offsetting GHG emissions with mitigation (i.e., reducing emissions) and sequestering C, rather than securing off-farm offsets.

Innovations and Research in U.S. Beef Production Technology Needs

The U.S. beef supply chain provided more than 12 million tons of meat to American and global consumers in 2019 (USDA, 2019b) and is the most segmented animal agriculture industry. The beef supply chain comprises seedstock producers, commercial cow-calf producers, yearling/stocker or backgrounding feedlot operators, finishing feedlot operators, packers, retailers, food service distributors, and consumers. While cropland agriculture has consolidated over the past decades, consolidating livestock is less persistent and, in some cases, very little changed from previous decades (USDA, 2020a). The 2017 Census of Agriculture counted 726,046 farms with beef cows, with a mid-point of 120 cows per farm. Forty-three percent of these farms have 10,000 acres or more of pasture and rangeland, but most have fewer than 500 acres (USDA, 2020a). For the most part, these extensive beef production systems graze cattle on pastures and rangelands. Much of this land is unsuitable for crops, so beef cattle provide a viable source of agricultural production and income (Steiner et al., 2014). While using arable land allows producing high-quality, human-edible protein, seedstock producers (breeders), cow-calf producers, and feedlots account for approximately 70-80% of the industry's total C footprint (Rotz et al., 2019). Consolidation of the production of pork and poultry, as well as intense consolidation of meat processing, has occurred across animal agriculture (Saitone et al., 2024). In this section, we focus on increasing C storage and sequestration, which are more applicable to extensive operations. Ways to mitigate emissions from intensive finishing feedlots focus on enteric methane, manure, and feed crops; these are the same challenges faced in the dairy sector and will be discussed in the dairy section. Reductions of GHG in feed are addressed in Chapters 2, 3, and 4.

Grazing Systems

Rangelands store up to 20% of the world's global soil organic C and cover more than half the terrestrial surface (Conant, 2012). In the U.S. rangeland and pastureland cover

163.4 million ha (21%) and 49.3 million ha (6%) of land area, respectively (Spaeth et al., 2020). Recently, their potential to increase soil C has made rangelands and grasslands a research focus (Sanderson et al., 2020). Rangeland types are tremendously diverse, varying significantly based on climate, weather, temperature, and soil type. The Western Great Plains comprise a significant portion of the rangelands in the U.S. and are well studied. In semi-arid rangelands, like the Western Great Plains, long-term (decades to centuries) drivers of soil C (defined as total organic C) are climate and soil texture, while weather and associated events (fire, flood, drought) are short-term (within year to several years) driver of changes in flux, irrespective of management type (Sanderson et al., 2020). Most (80-90%) of this ecosystem's organic C is in the soil (Derner et al., 2006), and much of that (85%) is highly stable unless disturbed. Beef production provides economic value to rangelands, keeping the ecosystem services intact. Thus, avoiding the loss of rangelands by conversion to other uses is important to preserve C stocks in soils and allocate carbon sequestration in range systems to beef systems (Derner et al., 2006; Sanderson et al., 2020). Previously cultivated or degraded lands provide additional opportunities to increase soil C (Rowntree et al., 2020; Sanderson et al., 2020). Restoring perennial vegetation and appropriately managing livestock grazing are critical to increasing soil C during restoration (Milchunas and Vandever, 2013; Fuhlendorf et al., 2002). Finally, adaptive livestock management, where livestock numbers are adjusted to fluctuating forage production, thereby matching use with plant production, improves increases in long-term soil C (Sanderson et al., 2020). The mechanisms behind soil C sequestration are not fully understood, but excessive stocking rates negatively affect soil health, increasing erosion, undesirable forage species, ultimately reducing the storage capacity and permanence of stored C (Sanderson et al., 2020). The most important practices that increase long-term soil C stock in rangelands and contribute to GHG negative beef systems are 1) avoiding conversion, 2) restoring cultivated or degraded lands, and 3) practicing adaptive livestock management.

We need more research and innovation to show how grazing management (i.e., adaptive livestock management) and differing forage species affect the ability of rangelands to store C (see Joyce et al., 2013, for example). To sequester more C in grasslands, the particulate organic matter pool (plant inputs) must increase at a rate greater than the rate of decomposition (see Chapter 3) (Cotrufo et al. 2019). Therefore, management strategies that increase above-ground productivity may benefit soil C sequestration, although this additional C may be more susceptible to loss via disturbance (Xu et al., 2018; Cotrufo et al., 2019). This has led to research into the potential of including legumes in grazing systems to increase animal productivity and soil C through increased N deposition and above-ground productivity. Soil C and N cycling have established relationships, but using legumes or fertilizer to increase soil N must be balanced against potential negative consequences, including increased nitrogen emissions and N leaching. These dynamics are addressed in Chapters 3 and 4 in more detail. Henderson et al. (2015) modeled net GHG mitigation potential in grazing systems and found that legumes may offset up to 28% of global C sequestration benefits. Developing and/or increasing the use of other highly productive forage species may help reduce N loss. Long-term sampling can help detect changes in soil C, but the considerable spatial and temporal heterogeneity associated with soil sampling makes this kind of research challenging.

Conventional life cycle assessment considers soil C in grazing systems at equilibrium but fails to account for C sequestration. In their review of research, Conant et al. (2017) reported that improved grazing management increased soil C sequestration by 0.28 Mg ha^{-1} yr^{-1} (one Mg is 10^{-6} Tg). Similarly, in a meta-analysis of comparative life cycle assessment (studies comparing regionally conventional vs. regionally improved beef production systems), studies that took C sequestration into account versus those that did not showed a 46% reduction in net GHG emissions per unit of beef produced (Cusack et al., 2021). Moreover, the meta-analysis showed that intensive rotational grazing and integrated field management in the U.S. reduced beef GHG emissions the most compared to any other region, although the number of peer reviewed studies was small. Research on the impact of grazing on other ecosystem services (e.g., water quality, wildlife habitat, erosion control), as well as C sequestration, should help ensure the sustainability of the industry. However, other than reductions in enteric CH_4 emission rates, innovations in soil C accumulation will be critical if the beef industry is to be carbon negative in the next few decades, the time frame for this assessment.

Innovations and Research in Dairy Production Technology

Milk production is the third largest agricultural industry in the U.S., which has seen tremendous improvement in efficiency and milk yield (per cow) over the last century. These gains came through investment in technologies, specifically genetics, nutrition, and animal management. Science and technology have allowed the dairy industry to intensify operations, so farms with more than 500 milking cows accounted for 63% of the milk supply in 2012 (USDA 2013a). The trend has increased in recent years with dairy farms in the U.S. declining by more than 50%, from 70,375 in 2003 to 34,187 in 2019 (USDA, 2019a) while milk production per cow has increased. The economies of scale combined with increased technology and improved herd genetics have contributed to the overall reduction of GHG intensity of U.S. milk.

Livestock production in general, and ruminants in particular, have been scrutinized for their impact on the environment, including C emissions to the atmosphere (Steinfeld et al., 2006). Dairy production contributed 31% of GHG emissions from livestock in 2018 (EPA, 2020).

Although the main source of emissions was enteric fermentation (25% of all enteric emissions), dairy production contributed an even larger proportion of methane from manure (52% of all manure methane emissions) (EPA, 2020). However, the broader context shows that dairy production contributes only 1.14% of all emissions in the U.S.

The main sources of emissions from dairy production can be categorized into feed production, the animals themselves (enteric and manure methane emissions), and farm management. Naranjo et al. (2020) conducted LCA of the California dairy industry and reported that feed production contributed 15% of farm-related emissions, enteric emissions contributed 38%, manure 41%, and farm management 5%. However, nationally, enteric methane emissions contribute more than manure to C emissions in dairy production (Thoma et al., 2013). Innovation in technology and research needs in the following three sections cover major emission sources. These emission sources are the same as those faced by beef feedlots and should be considered as similar processes.

Enteric methane emissions

Methane is a natural by-product of microbial fermentation of carbohydrates and amino acids in the rumen and the hindgut of farm animals (Hristov et al., 2013). Dairy cattle emit enteric methane, which represents approximately 2-11% of the dietary gross energy they consume (Moraes et al., 2014). Reducing enteric methane emissions mitigates environmental pollution and can improve feed use efficiency. Enteric methane emissions are related to feed intake, fiber, and lipid content in the diet as well as methanogens in the rumen. Therefore, technologies to reduce methane emissions target improving fiber digestibility, increasing lipid content of the diet, or using feed additives to either inhibit methanogens or modify the rumen environment.

Gerber et al. (2013) suggested that improved digestibility through better forage quality could have low to medium effectiveness in mitigating methane in dairy cattle. Efficacy of lipids in reducing methane emissions depends on the form and level of supplementation, as well as the source and fatty acid profile of feedstuffs (Beauchemin et al. 2008; Eugène et al. 2008). In dairy cattle, Eugène et al. (2008) reported up to a 9% decrease in methane emissions with lipid supplements (average 6.4%); control diets averaged a 2.5% decrease primarily because of reduced dry matter intake. However, the amount of lipids that could be included in the diet is limited because lipids suppress feed digestion, and the cost is high.

Recent advances in inhibiting methanogenesis can reduce enteric methane emissions dramatically. For example, Keraeb et al. (2023) reported that supplementing feed with 3-nitrooxypropanol (3NOP) at an inclusion rate of 0.123 g/kg dry matter reduced enteric methane production by 39% in dairy cattle. Roque et al. (2019a) observed reductions in methane intensity up to 67.2% (g/kg milk produced) in lactating dairy cattle fed diets supplemented with the macroalgae *Asparagopsis armata* at an inclusion rate of 18.3 g/kg dry matter. Other supplements like Mootral (garlic and citrus extract) and SOP reduced enteric methane emissions by more than 20% (Roque et al., 2019b, Ross et al., 2020). Honan et al. (2021) provide a more detailed review of feed additives that reduce enteric methane production.

Manure methane emissions

Different livestock systems, using practices to manage manure, contribute to methane emissions. The most common method of storing manure in dairy and pork production systems is a lagoon, which contributes more to methane production than other manure management systems. Reducing storage time in lagoons, reducing manure temperature by storing it outside during colder seasons, and using alternative manure management systems such as dry composting effectively reduce methane emissions across dairy and feedlot production systems (Montes et al., 2013). Capturing methane in an anaerobic digester and combusting the gas also effectively reduces methane emissions, thus reducing total GHG impacts (methane can be more than 30 times more impactful in heat capture in the atmosphere than CO_2). Manure organic C and N content from digesters can be more readily available for microbial processes and reducing methane emissions following land application (Montes et al., 2013). Acidification and manure additives like SOP Lagoon reduce methane emissions by more than 20% (Peterson et al., 2020).

Feed production

Crops and forages are essential as feed for most animal production systems, including dairy and beef cattle. Using manure from animals to fertilize crops increases circularity in the carbon budget, reducing GHG emissions across both systems. To reduce the C footprint, particularly reducing methane emission and enhancing soil C sequestration, dairy production can capitalize on genetics and management practices that improve forage and grain production and nutritive value (Martin et al., 2017). Improvements to forage crops increase fiber digestibility, crude protein, and non-structural carbohydrates (Brummer et al., 2009). Dicotyledonous plants have highly lignified secondary cell walls that become less digestible as the plants mature, which makes them the main impediment to digestibility (Jung et al., 2012) and contributors to enteric methane emissions. Dairy and beef cattle can better use alfalfa by (1) improving alfalfa to contain less lignin and (or) altering lignin composition for better fiber digestion; (2) breeding legumes to produce less tannin in leaves, which reduces enteric emissions; and (3) inserting genes into alfalfa to add polyphenol oxidase and o-quinones from red clover (McCaslin et al., 2015).

In some situations, dairy cattle spend time grazing pastures. Livestock produced under managed grazing systems can improve soil health, increase C sequestration, and potentially be a C sink, not a source of C emissions (Uddin and Kebreab, 2020). C sequestration was further discussed in the beef cattle section.

Innovations and Research in Pork Production Technology

The U.S. is the largest global exporter of pork (FAO STAT, 2020) with production increasing steadily over the last several decades. The general trend shows increasing consolidation and specialization in operations from pasture-based farrow-to-finish operations to separate piglet production and feeder pig finishing operations. Most U.S. pork is produced in the Midwest, Southeast, and Northern Great Plains (USDA, 2013). In the U.S. almost 20% of adults consume pork (Penkert et al., 2021).

The cradle-to-farm gate C footprint of U.S. pork ranges from about 2.5 to 3.5 kg CO_{2e} per kg of hog live weight (Pelletier et al., 2010; Putman et al., 2018; Tallaksen et al., 2020). Feed production and manure management both contribute heavily to pork's C footprint for both commercial and niche swine production systems (Tallaksen et al., 2020). However, region also influences C footprint results with warmer regions driving greater nitrogen dynamics, including microbial transformation and ammonia volatilization (Putman et al., 2018). Emission sources from feed production include fossil fuel combustion for tractors used during planting, harvesting, and storing feeds and nitrous oxide emissions from fertilized fields. Manure emissions, particularly methane, are mostly the product of anaerobic fermentation during liquid manure storage.

One opportunity for reducing the C footprint of pork to net-zero or net negative is by reducing and ultimately replacing fossil fuels with renewable energy sources and manure management (See Chapter 6). On-farm solar and wind energy can offset grid power demand, further reducing GHG emissions associated with animal production. Additional technologies for housing and managing swine also show promise in reducing GHG emissions. For example, hoop barns (low-cost polyvinyl-covered structures) used in an Iowa production system reduced non-renewable fuel demand by 63 to 64% with a corresponding 35% reduction in GHG emissions per hog; this also reflects reduced manure emissions, despite increased feed requirements (Lammers et al., 2010). Generally, solid manure management systems reduce the C footprint of pork production, although this comes with the significant tradeoff of increased eutrophication of emissions (Lammers et al., 2010; Pelletier et al., 2010).

Feeding decisions, like using byproduct feeds, may also contribute to net negative emissions by reducing crop production emissions. For example, replacing a portion of corn and soybeans in the finishing ration with dried distillers grains with solubles (DDGS) reduced the C footprint of grow-finish swine production by 2.7% (Stone et al., 2012), although the effect of including DDGS is not uniform. A retrospective analysis of U.S. swine production estimated that increased GHG emissions and energy use required to dry DDGS resulted in a bigger overall C footprint (Putman et al., 2018). Reducing pork's C footprint using DDGS as a byproduct depends on whether DDGS processing emissions are allocated to the ethanol or the DDGS themselves.

Supplemental feed ingredients like synthetic amino acids or phytase may also reduce feed requirements and thus also reduce the contribution of crop production emissions to pork's C footprint, but this may come with increased manure emissions because manure solid content is more volatile (Kebreab et al., 2016). In addition, the GHG emissions associated with synthetic amino acids are greater than other feed additives and plant-based protein (Benavides et al., 2020). The advantage of synthetic amino acids is the ability to fine-tune feed rations to support animal health and growth at each life phase, thus increasing feed use efficiency across all feeds. Facilities contribute a relatively small proportion of the farm-gate C footprint. However, swine producers have more control over reducing emissions associated with this phase of production by reducing electricity requirements using energy efficient technologies (e.g., replacing heat lamps with electric heat mats for piglet creep spaces, reducing barn temperatures at night, and environmental control systems that regulate fans and heaters; Tallaksen et al., 2020).

Innovations and Research in Poultry Production Technology

Poultry production has grown in recent decades to make poultry the most commonly produced meat worldwide (FAO STAT, 2020). Poultry production, compared to ruminant production, is more efficient at converting consumed feedstuffs into live weight gain resulting in a smaller C footprint (MacLeod et al., 2013), although its net protein contribution is less on a per kg feed basis (Ertl et al., 2016). Feed production is the largest contributor to the C footprint for poultry meat and eggs. Putman et al. (2017) estimated that broiler feed accounted for approximately 65% of the total global warming potential in poultry production. Similar results were reported by Pelletier (2017) for Canadian egg products. Pelletier et al. (2013) also conducted an LCA of Midwestern U.S. egg production and found feed production was the largest contributor to the poultry C footprint. Moreover, nitrogen losses from poultry manure contribute significantly to the C footprint (Pelletier et al., 2014). Since 1965, poultry production has reduced emissions with lower emissions per 1000 kg of poultry meat and per kg of egg produced by 2010. Total industry effects increased because of increased poultry production (Pelletier et al., 2014; Putman et al., 2017). Both comparative LCAs found that improved emission intensity was due to improved crop yields and increased bird performance (Pelletier et al., 2014; Putman et al., 2017).

The industry must balance feed rations with a lower environmental cost while maintaining improved bird performance to meet a GHG neutral or negative goal (Putman et al., 2017). Specifically, reducing animal-derived materials or including less ruminant-derived feed products, and other less GHG intensive derived material in poultry feed has been highlighted as an efficacious strategy (Pelletier et al., 2013). Meeting this goal will require diet formulation methods like those in Heidari et al. (2021), where regional environmental life cycle inventory information was incorporated in feed formulations. Such a tool should include the environmental impact of feed, nutrient content that supports the desired performance level, and the ration's cost. Nguyen et al. (2012) used a similar LCA approach to optimize rations. While such an approach is feasible, rations with reduced environmental footprints cost more, which agrees with similar efforts in dairy cattle (Moraes et al., 2015). While these approaches are promising, they require regionalized benchmarking of production systems. For further discussion of improving cropping systems, see chapters 3 and 4.

Improvements in bird performance and feed efficiency must occur while also meeting consumer demands for shifting management, such as cage free and free range eggs and poultry. Continued research in genetics, welfare, nutrition, antibiotic alternatives, and alternative production systems can help meet these goals (Putman et al., 2017; Gadde et al., 2017). Improving feed digestibility through genetic selection is a viable option for reducing N waste and GHG emissions. De Verdal et al. (2013) observed that genetically selecting broilers for improved wheat digestibility could decrease nitrate excretion by 13% compared to broilers for low digestive efficiency. However, while genetics is responsible for 85-90% of improved efficiency in the poultry industry, these strides may come at the expense of animal welfare (Zuidhof et al., 2014; Torrey et al., 2021). Torrey et al. (2021) found that conventional chicken strains reached a target weight of 3.2 kg faster than "fast", "moderate", and "slow" growing genetic lines and did so without affecting overall mortality rates. However, Dixon (2020) and Rayner et al. (2020) examined welfare indicators of slower and faster growing broiler genetics and found that slower growing broilers had better welfare than faster growing broilers. In laying hens, Grebey et al. (2020) observed that different laying hen strains showed genetic dust bathing and inter-bird differences that affected the success of potential stocking rates in aviaries. Continued benchmarking of genetic differences should help the industry balance calls for improved welfare and shifting production systems while reaching a GHG negative target.

Innovations and Research in Sheep and Goat Production Technology

Sheep and goat populations in the U.S. are smaller than other ruminant species, at 5.17 million sheep and 2.58 million goats in 2021 (USDA, 2021). Thus, these species have a lower environmental impact than other livestock species, although they will require emissions mitigation as regulatory and market pressures increase (Dougherty et al., 2019a). Like other ruminant species, sheep and goats are in highly diverse production systems with a C footprint largely driven by enteric methane production (Dougherty et al., 2019b). Mitigation strategies for reducing enteric methane are detailed in the dairy production section. However, sheep are a common model for exploring enteric methane mitigation strategies. However, we have little benchmarking of C footprints for small ruminants, unlike beef and dairy cattle (Jones et al., 2014; Roma et al., 2015; DeLonge et al., 2016; Dougherty et al., 2019b). Compared to large ruminants, the literature shows a data gap for small ruminant production, particularly in feed composition, breed-specific data on production and performance, and extensive production systems (Dougherty et al., 2019a). This reveals a knowledge gap in environmental and economic tradeoffs among management strategies and regional differences in production efficiency (Dougherty et al., 2019a).

In the literature, as with beef production, breeding stock contributes the most to the overall C footprint. Improving production efficiency has reduced the C footprint of the final product (Dougherty et al,. 2019b; Jones et al., 2014). Therefore, management that affects breed stock animals influences the production C footprint. Dougherty et al. (2019b) and Jones et al. (2016) conducted an LCA of California and U.K. sheep production systems, reporting that ewe replacement rate, lamb growth rate, and increasing lambs per ewe affected the size of the C footprint significantly. The same is likely true for goats. Improved efficiency correlates with a reduced C footprint across multiple species (Dougherty et al., 2019a). Management strategies that can improve production efficiency in sheep and goat production include improved nutritional efficiency through grazing management and forage quality, genetic selection for efficiency, radio frequency ID tags for data collection, and improved animal health and welfare. Continued research should also help improve nutritional models for small ruminants and improve performance in extensive systems (Cannas et al., 2019; Dougherty et al., 2019a). However, we are unaware of any published LCA for goat production systems in the U.S., although other regions have seen such studies. Filling this knowledge gap is necessary if we are to understand better how these industries can reach a GHG negative goal. In the U.S., benchmarking in sheep and goat production lags behind other livestock species. To better identify areas for improvement, tradeoffs between both economic and environmental management strategies, and track performance, we do require benchmarks (Dougherty et al. 2019b). Additionally, research on nutrition models specific to these species and extensive production systems should help accurately model these systems.

Potential for Improving Animal Protein Yield

Beef production efficiency has improved in recent decades as forage quality has improved, feed concentrate use has increased, and genetic selection has improved growth, reproductive performance, and animal health and welfare (Capper, 2007; Legesse et al., 2015). Because of these improvements, more meat is produced from fewer animals. A key area for improvement, however, lies with the supporting cow-calf sector because 70-80% of the beef industry's C footprint comes from these animals (Rotz et al., 2019). Further improved productivity may further reduce emissions per unit of beef produced.

In the U.S., while dairy cattle emissions increased 4.6% between 1990 and 2018, the dairy cattle population has declined by 2.6%, with milk production increasing by 57% (USDA, 2019). These trends indicate that while methane emissions per head are increasing, emissions per unit of product (i.e., meat, milk) are decreasing. Milk production has more than kept up with population increase; the average annual rate of milk production per cow increased 10.6% from 2010 to 2020 (USDA 2020b). Methane intensity decreased 52% from the 1950s (von Keyserlingk et al. 2013) and 45% from the 1960s (Naranjo et al., 2020). Milk production per cow should continue to increase, which may translate to fewer animals and thus further reduced methane from milk production.

The greatest opportunity for animal production for reducing GHG emissions lies in improving feed use efficiency through improved diet formulation and animal genetics, and reducing GHG produced in feed production (Tallaksen et al., 2020; Putman et al., 2017). However, to meet GHG negative goals, diet formulation must include environmental costs, renewable fuel alternatives in barns and feedmills, and enhanced animal production efficiency while maintaining high animal welfare standards.

The biophysical reality is that animals are carbon consumers and emitters, and do not sequester carbon. Primary producers are the only natural carbon sequesterers on Earth. The goal for animal production in achieving GHG negative agriculture is to reduce emissions and return carbon back to the soil as efficiently and effectively as technologically and economically possible. Animal proteins are a critical part of the U.S. and global food chain, creating high quality food from low quality feed that in most cases cannot be digested by people. When combined with effective GHG reductions in feed and increased carbon sequestration in soil, it is possible for animal agriculture to contribute to a GHG negative agricultural system in the U.S.

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Chapter 6:. The Challenges and Opportunities for Energy and Energy Use Efficiency

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Energy Use in the U.S.

Agricultural production is energy intensive. Harnessing energy from beasts of burden expanded agricultural production for over 8,000 years to feed a growing world population. Mechanized agriculture using geochemical energy beginning in the late 19th century created an unprecedented expansion in yields from the land and formed the basis for a global food supply chain that feeds eight billion people daily.

Agricultural production amplifies the ability of plants to convert phytochemical energy (sunlight) into biochemical energy (e.g., sugars, fats, proteins). The technological breakthroughs of the 20th and 21st centuries transform fossil energy into food (Conforti and Giampiertro, 1997), meaning abundant, affordable, and reliable energy sources remain critical for modern agricultural production. The U.S. Energy Information Administration Monthly Energy Review for January 2022 reported total annual U.S. energy consumption in 2020 at approximately 99 exajoule (EJ)s or just under 93 quads (quadrillion BTUs, U.S. Energy Information System, 2023) http://www.eia.gov/mer). The U.S. produced more than 95 quads of energy that year, making the U.S. a net exporter of approximately 3.5 quads of energy. The total annual energy use by agriculture as a sector is about two percent of the total energy consumed annually in the U.S., or 1.9 quads. Today most energy consumed to support agriculture comes from fossil fuels.

The large quantities of energy used in agriculture increase yields and decrease labor per unit area of production. Mechanized agricultural production has reduced human labor demands from approximately 1,200 hrs/ha for corn using animal power to 12 hrs/ha (Pimentel, 2019). This efficiency translates to a 100-fold decrease in encumbered energy to support human labor (food, housing, transportation).

Thus, the challenges of increasing net agricultural production to feed a growing human population while reducing greenhouse gas (GHG) emissions are profound (USEPA, 2021). These challenges present a tradeoff between acute risks of human suffering from nutritional stress and the chronic risk of changing climate conditions affecting production stability and capacity. This chapter explores energy sources in agriculture, potential renewable alternatives to meet energy demands, and potential energy efficiency gains through technological innovation.

Energy Use in Agriculture

Accounting for energy used in agricultural systems is difficult, in part because the system elements that make up the food supply chain are large, integrated across every economic and geopolitical sector, and distributed across the entire planet. The Farm Energy Analysis Tool (FEAT) developed by Pennsylvania State University provides a whole farm static model of farm energy and GHG emissions for different farm systems (Farm Energy Analysis Tool, 2023).

The most apparent energy use in agriculture is fuel for cultivation, harvesting, transporting, drying, and other on-farm activities. This direct energy use requires diesel fuel, gasoline, propane, natural gas, grid-sourced electricity, and renewable energy like wind, solar, and biofuel energy. Indirect energy includes encumbered energy and associated environmental burdens required to manufacture products and equipment to produce crops and animals. Energy systems inputs and outputs for agriculture flow through crop production, livestock production, on-farm energy production from fossil fuel, and on-farm renewable energy production (Figure 1).

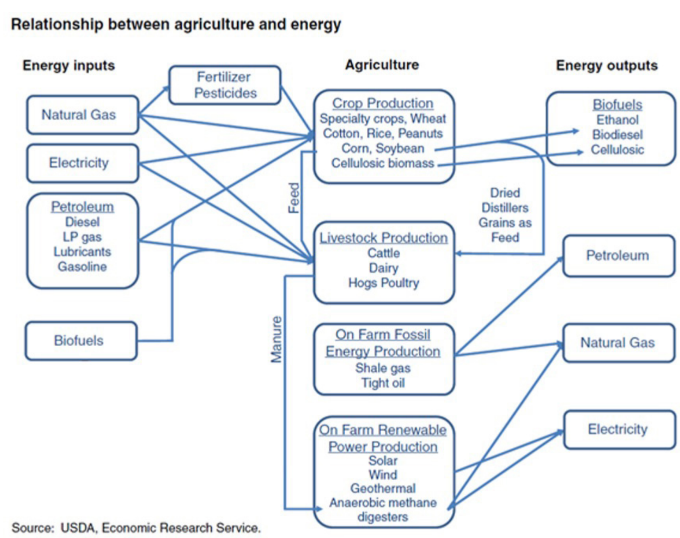

Figure 1

The nearly two percent of total U.S. primary energy consumption used by agricultural production drives more than $1.1 trillion in economic activities that comprise the U.S. food supply chain, or more than five percent of US GDP in 2019 (USDA, 2022). Of course, the total food supply chain includes transportation, processing cold-chain and dry storage, distribution, and consumption. The total food supply chain in the U.S. utilizes about 17 percent of the nation's fossil fuel energy (Pimentel, 2019). The necessity of a viable and resilient food supply chain for national security and domestic welfare has become increasingly apparent in the face of pandemic-driven supply chain disruptions. The use of energy by the agricultural sector is efficient and effective at leveraging the vastly larger phytochemical and biochemical energy flows that provide society with the food, chemicals and materials produced by the bioeconomy.

Direct energy consumption on farms is mainly diesel (44 percent of direct energy consumption), electricity (24 percent), natural gas (13 percent), gasoline (11 percent), and liquefied petroleum gas (7 percent) (Figure 2). Farm machinery is powered by diesel and gasoline. Irrigation, cooling, and lighting are predominantly powered by electricity, with natural gas and LP gas used in heating and grain drying. Indirect energy consumption from natural gas is required to manufacture fertilizer and pesticides. Direct energy is responsible for 60 percent of agricultural energy use and 40 percent from indirect energy consumption (Hitaj and Suttles, 2016). Agricultural producers can exercise discretion over direct energy sources when alternatives are available. However, they have very little control over indirect energy, except the decisions on the utilization of the materials and equipment so encumbered.

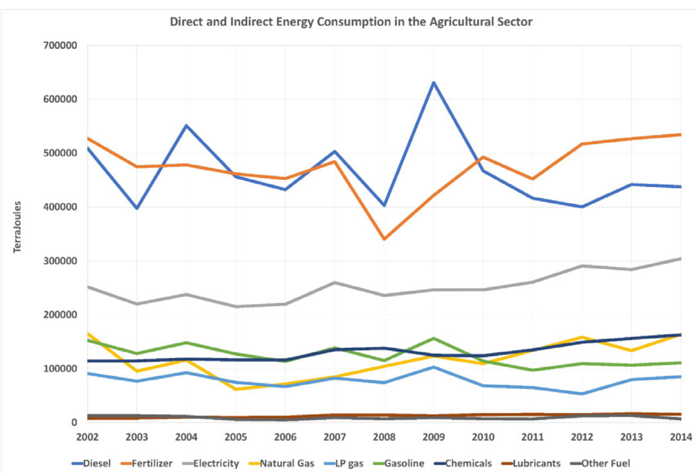

Figure 2

Reducing fossil fuel demands for producing crops requires a strategy that focuses on mitigating the highest energy demands: direct fuel, fertilizers, and irrigation water (Figure 3). For all non-leguminous crops, N fertilizer was the highest energy input for production, ranging from 3,175 MJ/ha-year (40% of the total) in hybrid poplar to 9,209 MJ/ha-year (43% of the total) in corn silage (Camargo et al., 2013), for legumes, on-farm energy use was the highest impact. The highest energy input demands averaged across all crops as a percent of total energy inputs were:

- Fertilizer - 36%
- On-farm fuel - 30%
- K2O - 7%
- Lime - 6%
- Transportation of inputs - 6%
- P2O5 - 5%
- Seed - 5%
- Herbicide - 4%
- Drying - 2%
- Insecticide - 1%.

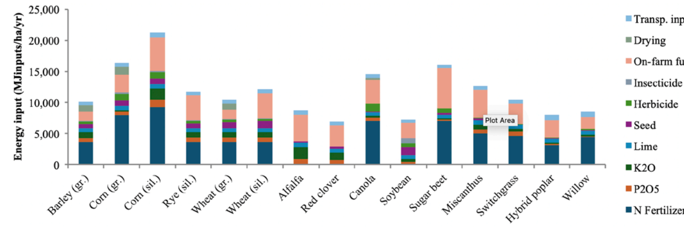

Figure 3

While on-farm energy use may not have the highest impact on many crops, reducing demand on fossil fuels while avoiding a yield penalty will increase producer economic resiliency and profitability. Increasing nitrogen use efficiency will reduce impacts from GHG emissions and nitrogen losses to the environment. Making decisions on energy systems within crops requires measures of production efficiency (energy output divided by energy input) and GHG intensity (GHG output divided by energy output). Efficiency metrics combined with impact metrics such as GHG intensity provide a more balanced understanding of the energy cost/benefits for each crop (Figure 4, Camargo et al., 2013). These analyses show that perennial legume, perennial grass, and short-rotation woody coppice crops were the most efficient, followed by annual silage crops, annual grain, and oilseed crops. Identifying where energy is most extensive in the agricultural production system provides perspectives on both the risks of disruption and opportunities for innovation in agricultural energy use.

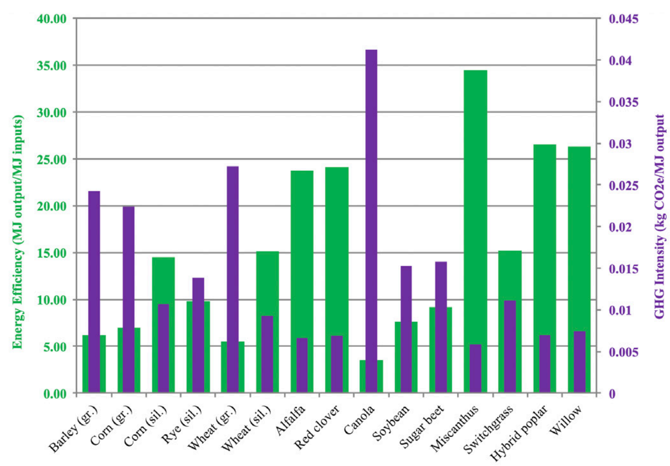

Figure 4

1. Energy Use in Machinery

The embodied energy in farm machinery is summarized in Table 1. Determining the energy requirements for machinery is complicated by the diversity of equipment uses, the schedules of use per crop, region, and season, and the efficiencies in terms of yields per hr of operation or kcal fuel consumed. Embodied machinery energy use per unit of food produced is also problematic. Direct energy efficiency (kcal fossil fuel plus embodied energy per kcal food produced) does not consider the nutritional density or value of the food. Food is produced for three primary nutritional characteristics (carbohydrates, fat, and protein), critical secondary nutritional characteristics (fiber, vitamins, and nutrients), and cultural values. Creating a single energy efficiency metric for food production is complicated by these factors.

The embodied energy in a single piece of agricultural equipment can exceed 50.24 MJ /kg. However, when amortized over the productive life of the equipment and the yield produced by the equipment, the encumbered life cycle impacts of machinery on agricultural production are almost always less than one percent of total encumbered impacts, based on yield, area, and time (Adom et al., 2012). This does not mean these encumbered impacts do not matter; they are important considerations in the energy budgets of agricultural production systems. However, the manufacturing and utilization of machinery is not a place to achieve significant gains.

Energy Type	Element	Energy (MJ/kg)
Embodied Energy	Tires	85.83
	Steel	6.28
	Tractor (Total)	49.46
	Combine (Total)	50.3
Fabrication Energy	Tractors	14.63
	Harvesters	13.0
	Primary Tillage	8.63
	Planters	8.63
	Secondary Tillage	8.35
	Sprayers	7.39
	Balers	6.28

Table 1

2. Energy Use in Fertilizers

Exogenous fertilization, especially nitrogen (N), phosphorus (P), and potassium (K), is critical for maintaining soil health, including carbon sequestration potential, while also supporting high crop yields. Energy used for fertilizers is highest in nitrogen. Energy inputs for the three most common nitrogen sources include anhydrous ammonia: 50.24 MJ per kg N, urea: 60.29 MJ/kg, and ammonium nitrate: 61.55 MJ/kg. The energy required for phosphate, including transportation, ranges from 2.5 – 6.3 MJ/kg and is 2.1 MJ/kg for potash (Pimentel, 2019). In addition to the embedded energy and GHG emissions associated with manufacturing, the dominant GHG emission from crop production in the field is nitrous oxide (N_2O), largely resulting from the microbial transformation of nitrogen fertilizers (Camargo et al. 2013). Thus, more efficient use of fertilizers can provide synergistic benefits from a system-level perspective.

3. Energy Use in Pesticides

Pesticides, including insecticides, herbicides, fungicides, and defoliants (for cotton), are composed of molecules derived from and synthesized with energy from fossil fuels. Almost half of the 600 million kg of pesticides applied to agricultural fields annually in the U.S. is herbicides, predominantly glyphosate and atrazine (USDA, 2022). Fruits and vegetables have the highest proportion of energy encumbered in pesticides from production (15 percent), with most agronomic crops encumbering less than five percent of total energy use by pesticides (Helsel, 2019).

4. Energy Use in Irrigation.

Energy costs for irrigation include the energy input to produce the irrigation equipment and piping (fixed energy costs) and the energy required to move water against the energy gradient (variable energy costs). The fixed energy costs are amortized over the functional life of the equipment and can be as high as 10.47/kg (assuming aluminum components) from (Pimentel, 2019). The energy cost of moving water is very high; water is dense (1 kg/l), and pumping energy gradients include head (height of pumping) and energy losses in pipe systems.

Renewable Energy Production and Uses

Land-based production systems are net energy capture systems. Agricultural and forestry production in the U.S. harvest 25.2 EJ per year of equivalent energy as biomass. Total fossil energy use in the U.S. is just over 100 EJ per year (USEPA, 2021). The agricultural and forest sectors in the U.S. capture and process more than 30 percent of total fossil energy use in the U.S. (Pimentel, 2019). For an annual growing season for corn (180 days, April – October in the U.S.) approximately 21000 GJ of solar energy is emitted on each ha of cropland. The corn plant converts 1.2 percent (251 GJ) of that sunlight to biomass, of which only 0.4 percent (79.5 GJ) is grain. Corn production requires about 21 GJ of fossil fuel energy per ha, or eight percent of solar energy input (Pimentel, 2019). These thermodynamic realities support the potential for agricultural systems to provide biomass energy that could be converted to biofuels to offset fossil fuel demands.

Production of biofuels, whether as a primary or secondary product, has been debated as a potential solution for off setting demands for fossil fuels. Concerns over indirect land use change impact from biomass production include emissions from the transformation of forest lands to agricultural production and other indirect impacts (Bhan et al., 2021; Gyamfi et al., 2021). However, the impacts of fossil fuel energy consumption are clear.

Liquid biofuel production is a rapidly maturing industry that holds the highest promise to decarbonize the transportation sector, especially shipping, aviation, and long-haul transport (Field et al., 2020). Renewable Natural Gas (RNG) is another rapidly growing bioenergy resource that now fuels over half of U.S. natural gas vehicles and can also provide dispatchable renewable electricity to complement intermittent renewables like solar and wind, as well as replace conventional natural gas uses for heat, power, and transportation (Skorek-Osikowska et al., 2020; Walker et al., 2018). While current RNG is primarily from livestock operations, where anaerobic digestion can significantly reduce the GHG emissions associated with manure management, continued growth will require converting crop residues and energy crops and thus compete with liquid biofuels (van der Zwaan et al., 2021; Parker et al., 2017; Jaffe et al., 2016). The challenges of increasing production demand from the land and all the associated encumbered energy inputs and impacts to produce biofuels are very complex. They require a clear understanding of the consequences of strategies on the global carbon and energy budgets and quantitative consideration of impacts on ecosystem services (Field et al., 2020).

Tradeoffs between GHG emissions reduction strategies and other global ecosystem services seem inevitable. The potential to increase soil organic carbon through improved land use management of marginal lands with perennial biomass production could provide both atmospheric carbon mitigation and increased ecosystem services (Field et al., 2020). Carbon dioxide byproducts from biofuel processing can be captured and sequestered using microalgae or in geologic formations, roughly doubling the carbon benefits of cellulosic biofuels and RNG (Alami et al., 2021; Xie et al., 2021; Field et al., 2020). Technological innovations in cellulosic biofuels production and expanded support for land use transformation will be necessary for the economic viability of these types of complex solutions.

Energy Efficiency and the Digital Agriculture Revolution

Agricultural producers are facing increasingly dynamic and complicated challenges in managing and mitigating risks associated with profitable production. The impacts of climate change on crop energy demands are significant, and increasingly variable costs of inputs, including fuel, fertilizers, and pesticides, are compromising the tools producers have to manage those risks. Seasonal shifts in soil heating, rainfall, soil moisture, maximum and minimum daily temperatures, and frost can dramatically alter the viability of cropping systems within a region. Some of these risks can be mitigated through energy use, primarily through tillage and irrigation. Some require enhanced pest management choices, especially improved and targeted crop genetics. However, these will not be adequate to secure the prosperity of agricultural producers in the near term.

Resilient crop production strategies that improve energy efficiency include high-resolution precision management of inputs and production processes (Wolfe and Richard, 2017). The combination of innovations in plant genetics, high geospatial resolution information technologies, and integration with automation and scale-appropriate precision agriculture technologies in the Digital Agriculture Revolution holds the potential to maintain yield improvements without driving up energy demands (Basso and Antle, 2020). Managed input controls with crop data feedback sensors hold the potential to optimize producer profit and reduce exogenous losses, including N losses through N_2O to the atmosphere and nitrates to the water (Figure 5). Advances in remotely sensed (RS) data acquisition, geospatial information system (GIS) analytics, and dynamic agroecosystem decisions support system (DSS) modeling has increased crop risk mitigation and production management effectiveness in the past decade. Integration of these technologies with soil and crop sensors, particularly soil moisture and crop water use sensor arrays, increases both nitrogen and water use efficiency (Wolfe and Richard 2017, Peng et al., 2020, Zhou et al., 2020). This approach could reduce N demand by as much as 36 percent in the U.S. Midwest. The benefits of increased N use efficiency are dramatic across energy use, GHG emissions, and water quality impacts. The avoided burden of CO_{2e} emissions from N fertilizer alone could be almost 900 kg/ha (Basso and Antle, 2020). Understanding and managing soil carbon dynamics at the sub-meter scale could enhance soil carbon sequestration, a major potential sink for anthropogenic atmospheric carbon (NASEM, 2019).

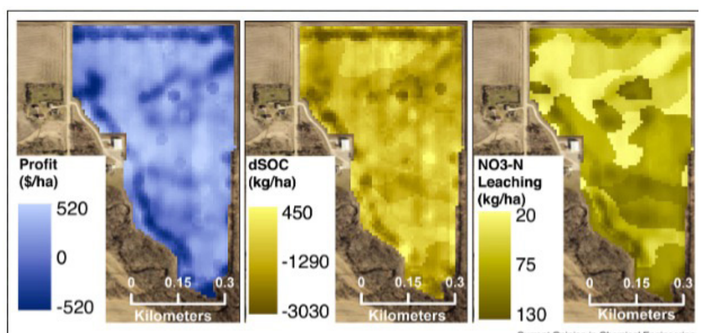

Figure 5

Conclusions and Recommendations

Future demand for food to feed a population of 10 to 12 billion people will require more energy input from either fossil or renewable fuel sources. Thermodynamic laws dictate that energy into a system will always be more than energy out of that system. However, Earth receives much more energy from the sun every day than is captured by photosynthesis. Biomass from sunlight is a persistent and unrealized opportunity for humanity. Transitioning the capture of biomass from fossil-fuel dependent energy sources to renewable sources will require a transition period characterized by focused effort on key transformational technologies.

This chapter has demonstrated that energy use in agriculture is integrated with nitrogen fertilizer and water use. Nitrogen use efficiency (NUE) improvements across cropping systems and the associated reduced emissions of N_2O, holds the highest potential of any single effort in reducing energy use and the encumbered environmental impacts of GHG emissions and water quality degradation from agricultural production (Allubione et al., 2011; Haroon et al., 2019). Water use efficiency (WUE), including improved irrigation controls and delivery technologies, crop genetics, and cultivation practices, will see similar integrated benefits, especially in areas that are experiencing climate-level shifts in precipitation patterns.

Expanding agricultural production to include perennial cellulosic biofuel crops produced on marginal lands to provide soil carbon sequestration will significantly reduce the net fossil fuel dependency of agriculture (Field et al., 2020). Finally, realizing the technological potential of precision agriculture will transform the levels of control and risk mitigation available to farmers at all scales worldwide.

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Chapter 7: The Challenges and Opportunities for Food Loss and Waste

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Global Food Loss and Waste

The global production and consumption of food pose one of the greatest threats to our environmental resources and the planet's wildlife. Agriculture production accounts for an estimated 70% of biodiversity loss (Grooten, 2021), 70% of freshwater withdrawal (Khokhar 2017), and 25-35% of greenhouse gas emissions (Tubiello et al., 2014), while 10 million ha of cropland are lost each year due to soil erosion (Pimentel and Burgess, 2013). As the global population and incomes grow across the developing world, food demand is projected to increase by more than 50% by 2050 (Searchinger et al., 2019).

Despite this staggering environmental cost, humans globally waste approximately one out of every three to four calories produced across the value chain from field to consumer (Lipinski et al., 2013). This food loss and waste (FLW) not only squanders environmental resources but also results in nearly $1 trillion in economic losses (Scialabba, 2015) and an overall loss of calories and nutrients for all consumers— particularly food-insecure populations. For instance, the U.S. spends an estimated $295 billion a year growing, processing, and transporting food that is never eaten (ReFED, 2021a). At a time when one-tenth of the world's population suffers from hunger or undernourishment (FAO 2021), global farm gate losses alone could feed the world's undernourished people approximately four times over (WWF-UK 2021).

FLW is also a significant contributor to climate change, producing an estimated 10% of global emissions (WWF-UK 2021) and 18-24% of the food system's GHG emissions (Poore and Nemecek, 2018; Springmann et al., 2018). That is four times the airline industry's climate impact, which is closer to just 2.5% of global emissions (Overton 2019). A significant portion of FLW emissions occurs at the farm level, with on-farm food loss alone producing 4% of all anthropomorphic GHGs and 16% of agricultural emissions (WWF-UK, 2021).

A key challenge FLW poses is that it accumulates incrementally via many disparate actors across the supply chain from farm to fork. To help address this complexity and coordinate greater action on FLW, the United Nations (UN) created Sustainable Development Goal Target 12.3 (SDG 12.3), which calls for cutting global food waste in half per capita at the retail and consumer levels, while also reducing food loss across production and supply chains. The UN has stated that by reducing FLW via SDG Target 12.3, countries can contribute to the Zero Hunger Challenge, improve their economic well-being, and reduce the food system's impacts on climate, water, land, and energy. The U.S. government has followed suit with a national goal to reduce FLW by 50% by 2030 (first set under the Obama administration and then continued under the Trump and Biden administrations).

U.S. Food Loss and Waste

Up to 40 percent of all food produced in the U.S. is lost or wasted (USDA 2020). Each year, 80.6 million tons of food is unsold or uneaten after it has been grown, processed, transported, or stored (ReFED 2021a). At the farm level, 17% of this surplus food (an estimate that only includes produce) is never even harvested (roughly 16.7 million tons) due to a combination of cosmetic standards, labor challenges, strict contracts, or neglect (ReFED, 2021a). It is estimated that much of it is still edible, up to 50% of it in the case of fresh produce (ReFED, 2021b). This is particularly tragic in the wake of the pandemic when as many as 1 in 4 American adults have become food insecure—with Black and Latinx adults affected at nearly three times the rate of white and Asian adults (Schanzenbach, 2020).

This high level of FLW creates significant climate risk for the U.S. and is now responsible for four percent of the country's GHG emissions—equivalent to emissions of 58 million cars annually (ReFED, 2021a.; US EPA, 2020). A quarter of these emissions result from the 13.9 million tons of crops left unharvested and largely unmeasured on farms yearly, while nearly 59% of FLW emissions occur in landfills and incinerators (ReFED, 2021a). Landfills are the third-largest source of methane in the U.S. and food is the largest input by weight into municipal landfills and incinerators (USEPA, 2014 and 2019). As a new UNEP report finds, cutting landfill methane emissions is one of the most cost-effective and critical steps to limit temperature rise to 1.5°C under the Paris Agreement (UNEP, 2021). Making matters worse, an estimated 80% of these municipal incinerators are in lower-income areas and on Native Nations' lands, disproportionately affecting underserved communities and communities of color (Li, 2019). Of particular importance to U.S. organic material and food waste going into landfills represents a loss of valuable organic matter and nutrients that, if recycled, could make it back to the farm to replenish soils, minerals, and nutrients.

To begin addressing these risks, the U.S. created the Federal Interagency Food Loss and Waste Collaboration—between the USDA, EPA, and FDA—to achieve a national target of halving FLW by 2030. By accelerating the U.S. Government's efforts to cut FLW in half by 2030, the nonprofit, ReFED, estimates the US can lower greenhouse gas emissions by 75 MMT CO2e annually upon implementation, rescue four billion meals for people in need each year, create 51,000 jobs over the coming decade, return nutrients to degraded soils on American farms and public lands, and boost profitability for American farmers and ranchers (ReFED, 2021a).

In 2015, the EPA developed the Food Recovery Hierarchy, which would become the backbone for organizations, businesses, and communities looking to reduce food loss and waste in America. The Hierarchy holds the important assumption that reducing or preventing the amount of surplus food generated in the first place will ultimately reduce demand for a product that is currently being wasted, in turn conserving more resources than donating or recycling food (Gunders & Bloom, 2017). The Hierarchy illustrates priority actions that organizations and businesses can take to prevent and divert wasted food, highlighting in order of environmentally beneficial preference interventions for organic waste: source reduction, feed hungry people, feed animals, industrial uses, composting, and landfill (See Figure 1.)

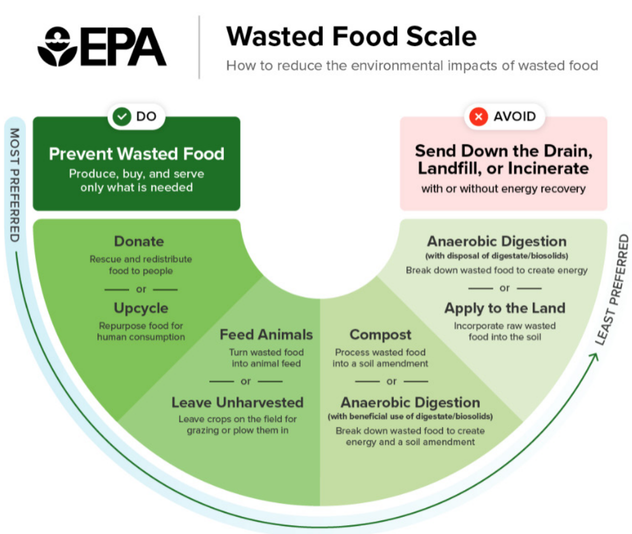

Figure 1: EPA Wasted Food Scale

Solutions to Limit Food Loss and Its Climate Impacts on Farms

Measurement and Metrics

Measurement is the first step in addressing new opportunities around waste reduction, financial benefits, environmental outcomes, and food and feed utilization. For instance, a recent WWF-UK report measured pre-harvest and harvest food losses on-farm and estimated that up to 40% of all food produced globally is lost or wasted, 1 billion tons more than previously estimated (10). On-farm measurement and data for loss at this stage of the supply chain is a huge gap and is often widely misunderstood by other actors along the supply chain. Specialty crop loss, for example, is highly variable for each crop type, region, and type of contract or market the product is sold into—which should not be estimated for all fruits and vegetables uniformly. Furthermore, there is a need for greater food loss measurement with staple commodity crops like corn, soybean, and rice, not only at the farm level but within animal feed systems to help optimize feed utilization.

Several sustainability measurement tools and metrics for growers exist in the U.S. today. However, there is only one tool for measuring food loss on farms specific to the U.S. and requires in-field measurement— the Stewardship Index for Specialty Crops' (SISC) Food Loss Metric. The SISC metric helps producers measure and understand how much edible and marketable food is left behind in their fields after harvest. This type of data can be used to identify current loss hotspots and downstream factors (such as buyer specifications, contracts, and other downstream factors) that drive loss today. The SISC Food Loss Metric is intended for growers to track and report the amount of food grown to maturity but not sold or donated; in other words, crops that were "ready for harvest" but did not enter the supply chain for human consumption. Understanding this amount of product "loss"—product left in the field or culled at various other stages in the supply chain—will not only provide growers with useful data but can allow for transparent information flows between growers and buyers that can potentially reduce on-farm loss through a host of solutions such as value-add processing, long-term contracts, and donation/diversion plans.

The SISC Food Loss Metric measures loss at each stage of a grower's operation, including fields, packinghouse/processing facilities, storage, and transport between each stage. This metric also tracks destinations for loss, drivers for loss, product loss excluding a change in moisture content, and opportunity cost for returning to the field to harvest products that would otherwise become lost. By institutionalizing and reporting loss measurement, growers can identify adjustment opportunities and improve operational efficiencies within their control.

As more farmers and commodity groups collect and share more data, farmers can benchmark themselves against regional and national averages to quantify their opportunity to reduce crop loss compared to their peer group. The Food Loss Metric attempts to tie the quantified losses to specific reasons – i.e., weather or pest damage, market dynamics, food safety, labor shortages, etc. providing growers and buyers with information that can inform future planting and management decisions. Measuring surpluses can help to identify adjustment opportunities to improve operational efficiencies. Identifying why food and commodity crops are not sold or harvested provides an opportunity to optimize the use of other resources and inputs such as water, chemicals, labor, and fuel.

It should be noted that SISC is not the standard globally. There are comparable metric platforms that exist in other parts of the world, such as WRAP's grower guidance and recording templates, the Sustainability Initiative for Fruits and Vegetables (which is setting a goal with European suppliers to cut food waste 25% in FLW for select commodities), and the Cool Farm Tool's suite of sustainability metrics for growers.

Proposed Solutions and Areas for Further Research

FLW is again not a point-source pollution problem but an accumulation of incremental waste across the supply chain, which requires collective action from all participants. Treating supply chains as a circular system makes it possible to not only design out loss and waste but also drive positive environmental and social impact. In addition, proven solutions exist to address this problem that saves money, time, and reduces GHG emissions.

WWF's No Food Left Behind research series and convenings have sought to bring together diverse supply chain stakeholders to explore actionable and systems-based solutions to FLW on-farm and across the supply chain (WWF, 2021). ReFED's Data Insight's Engine is the most comprehensive U.S. FLW database available, with stakeholder-specific cost-benefit analyses for 40 plus FLW solutions ranked by their economic, FLW, and emissions reduction potential (see Figure 2 below).

Figure 2: Top FLW Reduction Solutions by Annual CO2e Reduction Potential (ReFED, 2024)

ACTION AREA	SOLUTION NAME	EMISSIONS REDUCTION (100 YEAR)
	Consumer Education Campaigns	15.2M Metric Tons CO2e
	Centralized Composting	13.5M Metric Tons CO2e
	Portion Sizes	11.9M Metric Tons CO2e
	Manufacturing Line Optimization	7.87M Metric Tons CO2e
	Manufacturing Byproduct Utilization (Upcycling)	5.45M Metric Tons CO2e
	Meal Kits	5.41M Metric Tons CO2e
	Waste Tracking (Foodservice)	5.27M Metric Tons CO2e
	Standardized Date Labels	4.16M Metric Tons CO2e
	Markdown Alert Applications	3.93M Metric Tons CO2e
	Centralized Anaerobic Digestion	3.72M Metric Tons CO2e

A critical research gap in the U.S. is further studying the total amount of on-farm food and commodity loss across all crop types, and the potential of reducing that loss to mitigate habitat conversion (of grasslands in the Northern Great Plains, for instance) and agriculture's overall greenhouse gas emissions. Today, roughly 27% of all food loss and waste occurs on-farms, with 14 million tons never even harvested, leading to 1.7M MT CO2e (ReFED, 2021c). The nonprofit ReFED estimates that if the identified producer-focused food loss solutions are implemented (with sufficient private and public investment), it could reduce U.S. emissions by 853,007 MT CO2e per year (ReFED, 2021d). However, these estimates only account for specialty crops, and there is a clear need to institutionalize better measurement and reporting of row crop commodity losses as well. While row crop commodity loss in field, transport, and storage may be very low as a percentage of the total harvest, better measurement could uncover opportunities for loss reduction that could support additional yield and farmer profitability and bolster support for avoiding additional habitat loss and landscape conversion to row crops (i.e. GHG avoidance).

The following is an excerpt from our 2020 No Food Left Behind: Part IV:

Globally, co-/by-products from crop production and waste from food supply chains constitute nearly 30% of global livestock feed intake (Mottet and De Hann, 2017). Additionally, in the U.S., roughly 10% of surplus food (7.66 M tons) is already sent to animal feed (ReFED 2021). Roughly half (3.7 M tons) of this is coming from the manufacturing sector, with another large contingent from grocery retail (1.8 M tons). While animal feed is a leading end-of-life option for many agricultural sectors, roughly 14.7 M tons of food waste are still going to landfill. This contributes to the 20% of total U.S. methane emissions coming from waste management (US Environmental Protection Agency 2021) that could be going to a higher value use, such as animal feed (ranked third on US EPA's Food Recovery Hierarchy), which is an age-old practice that deserves renewed attention using 21st century technology and practices. It is important to note that total food waste generation estimates in the U.S. are much higher (more than 27.6 M tons annually in the U.S., ReFED, 2020), but WWF estimates that only 14.7 M tons could effectively be used for waste-to-feed pathways due to issues of post-consumer contamination and viability of the feedstock. These waste-to-feed pathways also have the potential to lower demand for commodity row crops that cause habitat loss and landscape conversion (i.e GHG avoidance).

In 2020, WWF completed an unpublished study to measure the post-harvest loss levels and key loss drivers for soybean and corn in the Midwest using primary data collection methods, which found that average field-level losses were 4.7% (8.8 bushels per acre) on corn farms and 4.5% on soybean farms (2.3 bushels per acre). Data was collected on 16 corn fields and 15 soybean fields. The field sample collection protocol and survey were developed based on the Commodity Systems Assessment Methodology (CSAM), a step-by-step methodology for describing and evaluating post-harvest losses that include interviews of value chain actors, observations of harvesting and handling practices along the chain, and direct measurements of quality and quantity losses along the chain. Applying these loss rates to national corn and soybean production would imply there could be as much as 816 and

201 million bushels of loss, respectively, each year. While further research is needed to validate these findings due to the relatively small sample size, it highlights a potential disconnect between actual measured loss levels and what farmers and extension agents often estimate—which is typically a 1% loss for corn and 3% loss for soybean of a farm's total yield (ReFED 2021c). In this recent WWF research, farmers estimated their losses would be closer to 1.2 and 1.56 bushels per acre, respectively. Yet, actual loss results measured in the field and storage were often higher. Even if loss decreased by a modest amount per acre, it could potentially reduce emissions at scale nationally. In addition to this potential for lowering direct emissions associated with corn and soybean production, it is also critical to note that further research is needed to understand the relationship between commodity loss, overproduction, and the conversion of native habitat—which also has clear climate implications. Institutionalizing the continuous measurement and comparison of commodity loss rates globally also provides valuable data and further incentive to invest in commodity loss reduction in other parts of the world where loss rates may be substantially higher compared to the U.S..

The following sections provide examples of promising FLW interventions well positioned to specifically reduce specialty crop FLW on-farm that require further testing and research.

Innovative Contract Practices

U.S. food retailers make purchases today based on forecasted demand and strict cosmetic specifications, which increases FLW and limits the profit potential of the entire supply chain. Yet innovative contract practices—such as longer-term contracts, whole crop purchasing, and dynamic pricing systems based on quality for excess product—offer an alternative path that would incentivize buyers and growers to work together to find new ways to use more of the total available produce.

Long term and whole crop contracts can help buyers and growers form closer partnerships that identify new ways to utilize edible and marketable produce that would otherwise be left behind. This might include investing in local processing, canning, or freezing facilities or increasing food donations and distribution capabilities. It can also work to improve communication and data sharing around demand forecasting, consumer trends, harvest, and waste data—all of which adds stability and flexibility to farmer operations. The U.S. grading systems for fresh produce is based on sizing conditions and ripeness. Retailers take that a step further to ensure that shelf-life and cosmetic appearances are perfect, leaving little wiggle room for the product in the field that is misshapen, discolored, or too big or too small. With longer-term contracts in place, growers may have the opportunity to coordinate value-added processing for the product that do not meet specs or on-going donation. Lastly, long-term contracts can offer farmers the time and stability to invest in climate resilience measures, which in turn helps buyers and retailers mitigate long-term supply-chain sustainability risks.

These types of innovative contracts are not new. In the U.S., long-term contract systems are already used with different commodities, such as dairy. Globally, they are employed by major retailers such as Tesco in the U.K.. Several successful examples of the benefits to Tesco and other international retailer suppliers include farmers sharing crop growth and harvest forecast data to help avoid overproduction and over-purchasing; growers implementing regenerative and climate-smart practices to reduce weather-related losses; and producers partnering with retailers and buyers to identify alternative channels (such as food donation, value-added processing, or imperfect produce product lines) for food that would otherwise have gone to waste (Clay, 2018).

By helping to use and distribute what grown, especially nutritious specialty crops, innovative contract systems such as long-term contracts offer a holistic solution to addressing many of the social and environmental issues in the U.S. produce supply chain, including poverty, FLW, racial inequity, and food insecurity understanding these types of contracts is a critical step in shifting agricultural production of specialty crops to build a better food system that is more diverse, inclusive, and resilient.

Pairing On-Farm Food Loss Measurement with Regenerative Agriculture

Reducing both pre- and post-harvest loss means driving efficiencies and cost savings for producers while reducing overproduction, conversion pressures, and inputs like fertilizers and freshwater that affect climate, biodiversity, and ecosystem health. Maintaining soil health is one way post-harvest loss can be reduced via improved plant health and resilience.

Additionally, with greater adoption of regenerative agricultural practices (e.g., soil testing, planting cover crops, low-till farming, and crop rotation), growers can play an integral role in supporting biodiversity conservation, soil health, and ecosystem services (such as cleaner water). Farmers stand to benefit from decreased input costs (by reducing fertilizer and irrigation usage) and potential crop losses (from droughts and extreme weather). Most farmers are interested in regenerative practices for these very reasons but have struggled to make the switch from business-as-usual due to concerns about the cost and complexity of transitioning.

With this growing interest in regenerative practices, it's important to examine how to integrate food loss measurement within regenerative agriculture efforts. This could include research to test and measure whether pre-harvest factors (e.g., water infiltration, abiotic/biotic stress, soil profiles) are reducing the yield gap and whether regenerative agricultural practices can mitigate these factors to help reduce loss and improve ecosystem services. Participating growers will need to first baseline and forecast their total yield potential and loss levels. This data can then be used to determine the increased environmental and ecosystem benefits of switching to regenerative practices and compare the input costs and usage of regenerative agriculture row crop farms to conventional farms and the environmental performance and loss levels of regenerative row crop farms to conventional farms.

Policy Solutions

In addition to the on-farm interventions highlighted above, federal policymakers require urgent action to ensure we meet our national goal of halving FLW by 2030. The USDA plays a leading role, with the EPA and the FDA, in the federal interagency effort to achieve this goal and start toward building a regenerative and resilient food system that helps mitigate climate change, reverse nature loss, and deliver positive outcomes for producers and consumers.

To help realize these benefits and drive widespread adoption of FLW solutions, federal policymakers should prioritize the following policy measures: 1) invest in infrastructure to measure, rescue, and prevent all organic waste from entering landfills; 2) expand incentives to institutionalize surplus food donation; 3) assert the U.S. Government leadership on FLW globally and domestically; 4) educate consumers via private and public food waste behavior change campaigns; and 5) require a national date labeling standard. Detailed recommendations can be found in the U.S. Food Loss & Waste Policy Action Plan for Congress & the Administration (Food Waste Policy Action Plan, 2021).

The USDA can take immediate action to accelerate FLW reduction efforts by: issuing guidance to states on optimal regulations regarding feeding food scraps to animals; launching an education campaign on food donation liability protection under the Bill Emerson Good Samaritan Food Donation Act; conducting outreach to train and develop the capacity of small and mid-scale growers and food groups to participate in online and direct-to-consumer distribution such as SNAP; investing in food hubs and creating regional supply chain coordinator positions to oversee the efficiency and adaptability of regional food supply chains (aggregating critical data sources on surplus products, stranded assets, and gaps in cold storage and distribution infrastructure); updating its definition of compost products so that a greater number of potential buyers (such as farms, golf courses, or other operations near waterways) are encouraged to purchase compost; developing a marketing campaign to build compost demand; and expanding programs such as the Community Compost and Food Waste Reduction pilot projects to generate new compost infrastructure—with an emphasis on making compost accessible to farmers.

The sections below provide a more detailed look at several of these policy recommendations to help reduce FLW and its associated GHG emissions across the agricultural sector.

Regional Coordination of Supply Chains

Nationally, less than 10% of food is donated rather than wasted (Yaffe-Bellany, and Corkery, 2020)). In 2020, the sudden demand shift at the onset of COVID-19 exposed the inflexible and siloed nature of existing supply chains when canceled contracts (in the restaurant and hospitality sectors, for example) left surplus food stranded on American farms, even as demand at food banks and grocery stores skyrocketed. To help farmers and ranchers become more resilient, profitable, and capable of donating food that would otherwise be lost, the USDA should prioritize developing resilient local and regional food supply chains by 1) empowering producers to sell through new direct-to-consumer distribution channels and 2) providing fresh and nutritious foods to the growing number of families facing hunger. The USDA has signaled its intent to explore this further through its Build Back Better (BBB) initiative (USDA, 2021a).

The contracting process for any such initiative needs to be transparent and explicitly inclusive of different scale growers, especially minority- and women-owned, small- and mid-scale, or organic operations, and local food operations. This should include conducting outreach to train and develop the capacity of small and mid-scale growers to access new channels of distribution, such as e-commerce and direct-to-consumer. These growers can be promoted in published lists of “USDA-encouraged” or “USDA-supported” producers for contractors to easily reference. Building on the support from Congress in the American rescue plan, the USDA should aid the rollout of technology that allows small-scale producers, independent retailers, farmers, and farmers markets to participate in online SNAP markets. Altogether, these efforts can improve food security among SNAP participants, create economic opportunities for small-scale growers, and bolster regional supply chains.

Additional programs and infrastructure already in place can also have a huge impact on strengthening regional supply chains and reducing FLW, such as food hubs, which should be scaled and strengthened. According to the USDA, food hubs are centrally located facilities facilitating the aggregation, storage, processing, distribution, and marketing of locally or regionally produced foods (NRDC, 2019). Food hubs increase the efficiency and adaptability of regional food supply chains by focusing on connecting available food to local businesses and food banks, advocating for value-add processing, and aggregating critical data points on quantities of surplus or available products on regional farms—which was evident in the early rounds of funding of the Farms to Families Food Box program during the pandemic, Policymakers should develop programs and funding to increase the role of food hubs in regional supply chains and specifically help them to: 1) boost their capacity to meet communities where they are and serve vulnerable populations suffering from food insecurity who might not otherwise participate in traditional food assistance programs; increase the quality and nutritional value of food products donated to food insecure communities; and 3) assist producers in identifying alternative distribution and marketing channels (such as value-added processing, food donation, or other secondary markets outside of traditional restaurant and grocery retail). Doing so can help to decrease the amount of food lost (and its associated emissions) across the value chain, address food insecurity, and strengthen regional supply chains against future shocks.

The USDA could additionally create new regional supply chain coordinator positions to oversee the efficiency and adaptability of regional food supply chains. Similar to food hubs, these positions could focus on identifying and aggregating critical data sources on surplus products, stranded assets, and gaps in processing, cold storage and distribution infrastructure. Without real-time food supply data, regional supply chains will remain inflexible and struggle to transport food from growers to those who need it most in the face of sudden demand changes as we saw during the pandemic. These positions should prioritize bringing federal funding and assistance to food deserts and work to strengthened networks of producers, distributors, food banks, community and faith-based organizations, and other nonprofits that provide food assistance.

Build Demand for Compost On-Farm

Beyond expanding the nation composting infrastructure (and its potential to mitigate emissions by 4.94M MT CO2e per year), policymakers must stimulate demand for finished composting products to help develop a more circular and low-carbon food system that keeps food out of the trash. To do so, policymakers should prioritize the following measures: 1) expanding the USDA definition of compost products to attract new buyers (such as farms, golf courses, or other operations near waterways); 2) developing a marketing campaign for compost products; and 3) streamlining the compost contract process to help match compost generators with potential buyers (NRDC, 2019). The Community Compost and Food Waste Reduction pilot projects, authorized in the 2018 Farm Bill, should also be reauthorized and expanded to develop further and implement municipal compost and food waste reduction strategies, emphasizing making compost accessible to farmers (USDA, 2021b).

Eliminate Barriers to Feed Food Scraps to Animals

Overly rigid restrictions and bans divert food scraps that are safe and wholesome for animal consumption into landfills. The FDA and USDA should issue guidance to states on optimal regulations regarding feeding food scraps to animals (ReFED, 2021e), which state legislators and agencies can use to review and revise their policies (Broad et al. 2016).

Enable Greater Food Donation by Farmers

Given that many farmers operate at low profit margins without sufficient tax liability to claim a tax deduction, USDA and Congress must develop an alternative to the current federal enhanced tax deduction for food. Doing so could help incentivize farmers to donate surplus crops, which is too expensive for them to do so today. An alternative enhanced tax deduction should therefore help offset these donation costs (harvesting, processing, packaging, and the transportation costs of donating agricultural products to local food banks) and be simple and easy to calculate.

Conclusion

As the climate changes and the food system shifts in increasingly unpredictable ways, reducing FLW as part of a government overall approach to mitigate climate risk and food insecurity will be imperative. FLW could become more of a priority when governments establish forest and grassland conversion-free goals to avoid habitat loss. Wasted food is not only expensive but also is an environmental tragedy because of the enormous loss of natural resources used to produce and transport food from farm to fork—with nearly one-fifth of U.S. cropland, fertilizers, and agricultural water used to grow food that is ultimately wasted (ReFED, 2016).

Addressing FLW starts with data collection and radical transparency across the supply chain to uncover FLW hotspots that inform reduction strategies. Promising on-farm FLW interventions, such as innovative contract practices (e.g., long-term contracts and whole crop purchasing) and integrating regenerative farming practices, will require further research and funding. Farmers often under-estimate their losses (Pearson, 2018), and helping them institutionalize measures and work to prevent on-farm food loss offers the potential to drive efficiency across the global food supply chain for the benefit of people, companies, and the planet.

At the federal policy level, the USDA, FDA, and EPA are already leading on an interagency effort to reduce FLW by 50% by 2030. However, the only way to achieve this goal is by implementing policies that accelerate the widespread adoption of FLW strategies (Food Waste Policy Action

Plan, 2021)—which can also reduce the climate impact of our nation's food system by 75 MMT CO2e annually. To address FLW in the agricultural sector, these policies should increase the transparency of loss measurement, invest in regional supply chain coordination; remove barriers for farmers to programs that oversee regional supply coordination; build demand for compost; eliminate restrictions on feeding food scraps to animals; and enable greater food donation by farmers.

Prioritizing FLW in new infrastructure and climate legislation will send a clear market signal to states, cities, companies, and other countries to similarly make FLW reduction an official part of their climate strategies—while it helps consumers and businesses to keep food out of the trash and reduce the food system emissions.

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Chapter 8: The Challenges and Opportunities for Economic and Policy Research

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Introduction

The current administration has been addressing the climate crisis with a renewed interest in climate policy and climate research to achieve the goal of net negative emissions for agriculture by mid-century. This policy goal is motivated by a voluminous body of research showing the potential impacts of climate change under projected trends in greenhouse gas emissions (GHGs), documented in U.S. Climate Assessments mandated by Congress. As yet, however, the United States lacks a process to design and implement the policies and programs that could achieve the goal of net negative emissions for agriculture. Thus, our goal in this chapter is to identify the economic and policy research challenges and opportunities to design and implement a technically, economically, and politically feasible pathway for a more sustainable agricultural sector that can also contribute to the emissions reductions goals for the U.S. economy, focusing on reducing net GHG emissions and increasing carbon sequestration to achieve net negative GHG for agriculture.

Research on agriculture's role and opportunity in climate policy and mitigating greenhouse gas emissions began in earnest more than 30 years ago. Research has provided measures of the impacts of a changing climate on agricultural productivity, quantified levels of current and past emissions, and addressed the role of agriculture in reducing greenhouse gas emissions. Antle and McCarl (2002) and Paustian et al. (2007) provide reviews of the early literature, with many other more recent reviews including the various IPCC reports and surveys such as Eagle et al. (2012) and EPA (2024).

What these surveys show is that, despite decades of research, much uncertainty remains over the *technical potential* for agriculture to reduce greenhouse gas emissions as well as *policy options* that can or should be used to achieve this technical potential and contribute to the net-negative goal. This uncertainty is due, in part, to the lack of investment in systematic development, testing and improvement of models in a rigorous framework such as the one developed for global climate models by the Coupled Model Intercomparison Project (CMIP 2024).

Sustained investment is needed to integrate climate science and technical potential for reducing emissions into the large body of knowledge on policy design generally and agricultural and environmental policy specifically. Technologies and innovations to reduce emissions will only have impact if they are adopted by the agriculture and food systems. Adoption in turn hinges on credible assessment of economic consequences and the effective economic incentives. Given the many diverse and competing interests within the agriculture industry and the food system, systematic economic research is needed to establish a scientific consensus on agriculture's appropriate role in climate policy design for greenhouse gas mitigation and for adaptation of agriculture to climate change. These two aspects are closely inter-related and provide the pivotal connection to defining and implementing sustained pathways for NNE. We also must recognize that climate change is part of the larger sustainable development challenge – to define and commit to a sustainable positive trajectory for human well-being within the limits of the natural world.

To achieve a scientific consensus on agriculture's role, we propose a coordinated, protocol-based, and system-based approach to design and evaluation of technology and policy options at the national level, linked to more detailed analyses at the sub-national level focused on major production systems. This type of approach to agricultural systems modeling was pioneered by the Agricultural Model Intercomparison Project starting in 2010 (Rozenzweig et al. 2013, 2018). In this chapter we identify key lessons learned from research aimed at developing a systematic, systems-based approach to the evaluation of agriculture's potential contribution to NNE, computational tools to evaluate the contributions that existing or prospective new technologies can make towards NNE, and the kinds of new research investments needed to inform policy design and implementation. Our key recommendations related to economic and policy research to address challenges and opportunities are:

- Extend the national-level assessments of climate impact to a framework for the evaluation of agriculture's potential contribution to the NNE goal. This assessment framework would be designed to envision future sustainable pathways for agriculture and the food system and assess potential impacts of key technologies by region and major system (major grain crops, vegetables, fiber, livestock, etc.).
- Use this new evaluation framework to assess the effects of alternative policy mechanisms such as those being proposed now for agricultural producers (e.g., incentives for adoption of climate-smart practices), in the context of broader national policies (e.g., national carbon tax, import taxes, government-supported carbon bank, etc.).

- Scale up the investments in the data and analytical tools needed to evaluate the contributions that existing or prospective new agricultural technologies can make towards NNE.
- Create a portfolio of policies (such as food labeling and certification) needed to link demand for sustainably produced food – including but not limited to foods that are climate-smart – through food supply chains to the incentives perceived by agricultural producers and evaluate essential verification and tracking standards for these policies to be effective pathways for long-term emission reductions.

In the final section of the paper, we discuss recent examples that illustrate the need for these research investments to evaluate how agriculture can contribute towards the NNE goal and summarize specific research needs.

Climate Policy History and Agriculture's Role

Research on the impacts of climate change on agriculture, and agriculture's potential role in achieving greenhouse gas emissions reductions, began in earnest in the 1990's. The Global Change Research Act of 1990 mandated that the U.S. Global Change Research Program (USGCRP) deliver a report to Congress and the President no less than every four years on the impacts of climate change on major sectors of the U.S. economy including agriculture. Unfortunately, its mandate was limited largely to impacts and thus did not result in a scientific consensus on the *actions* – including research on needed greenhouse gas mitigation technologies, and policies to support their adoption – needed to achieve a goal such as NNE. One of our recommendations is that this mandate should be extended to research a scientific consensus on the policy options for agriculture and the food system. A major study supported by USDA made a first step in this direction, but an ongoing research program is needed (Brown et al. 2015).

It is also important to recall that, based in part on the success of the cap-and-trade policy for control of air pollutants implemented in the United States by the 1990 Clean Air Act in the 1990s, the Clinton Administration encouraged the global policy community to use a cap-and-trade approach to the Kyoto Protocol, the first major international climate policy agreement. However, agriculture was largely excluded from the Kyoto Protocol due, in part, to what were perceived at that time as scientific uncertainties surrounding agriculture's potential to contribute to permanent reductions in greenhouse gas emissions through carbon sequestration in soils and biomass.

It is also important to acknowledge important policy setbacks that have occurred over the past three decades. After appearing to support the Kyoto Protocol during the 2000 election campaign, the Bush Administration refused to participate in the agreement. Both the Bush and Obama Administrations failed to implement a national climate policy that would aim to reduce emissions through mandated reductions or incentives such as a cap-and-trade system or a carbon tax. Moreover, in 2020, the Trump Administration formally took the United States out of the "Paris Accord" reached in 2015 that the Obama Administration had supported; in early 2021, the Biden Administration rejoined the accord.

However, the lack of a comprehensive national climate policy did not prevent state governments from advancing climate mitigation and adaptation programs. Notably the cap-and-trade programs created in California and the Regional Greenhouse Gas Initiative in the U.S. northeast incorporated some elements of agricultural mitigation to offset industrial emissions. Initiatives in the private sector created "carbon offsets," however, the lack of a coherent national policy limiting greenhouse gas emissions meant that a formal national "carbon market" for a standardized and certified unit of carbon did not materialize.

Despite the challenges to climate policy in the U.S. and globally, research on technologies for agricultural mitigation continued, as documented in the other chapters in this report. We now know that with the scientific advances that have occurred over the past 25 years, and with further investment in agricultural research guided by climate-related goals, there are many opportunities for agricultural production systems to reduce their own emissions and to offset emissions from other sectors of the economy, and thus contribute towards the NNE goal. Through the various climate policy initiatives that have been undertaken globally as well as in the United States, much has been learned about the kinds of policies that are both feasible and necessary to achieve substantial progress towards the NNE goal. *However, what is still lacking in the United States is a comprehensive, national effort to synthesize the technical and economic knowledge so that it can be used to inform the design and implementation of an effective climate policy that would exploit agriculture's potential contributions.*

What We Know: Technical, Economic and Policy Options

In Table 1, we summarize the main points from what we have learned about technical, economic and policy options for agriculture. From a sustainable development perspective, the research community has identified key indicators in the three dimensions of sustainable development – economic, environmental, and social. Indeed, the United Nations has identified 17 Sustainable Development Goals (SDGs) and indicators associated with them (https://unstats.un.org/sdgs/) and an annual Sustainable Development Goals report (https://unstats.un.org/sdgs/report/2020/), that includes climate goals and indicators as well as food security goals and indicators. Thus, the scientific, policy and civil society communities have established a normative framework both for expression of goals, as well as a positive framework in which to analyze progress towards those goals. This positive framework also provides a way for the research community to study how alternative mechanisms, including technological innovation, institutional change, and public policy may impact progress towards these goals.

Climate and agricultural science also have identified the key greenhouse gases that are of concern for agriculture. Carbon dioxide is the major greenhouse gas emitted from agriculture through land use change, including deforestation, from agricultural tilling of soil, use of fossil fuels, and so on. But also important are nitrous oxide emissions from the use of nitrogen fertilizer, and methane emissions from livestock, animal waste, and some cropping systems such as irrigated rice. These three major components of agricultural emissions, and the ways that production systems can be managed to change these emissions, have been extensively studied, as we can see from other chapters in this volume. In some cases, agricultural cropland, rangeland, and forests can be managed in ways that act as carbon "sinks" by sequestering or storing organic carbon in soils or in biomass for some period of time.

Another area of scientific advance over the past four decades has been in the development of computer-based models that can simulate crop and livestock growth, various environmental processes including the behavior of the key greenhouse gasses, and the global climate. These literatures are too vast to summarize here, but some recent papers have addressed, for example, agricultural systems models (Jones et al. 2017); the periodic reports by the Intergovernmental Panel on Climate Change summarize advances across the related literature. In addition, major advances in modeling economic systems and their inter-connections with ecosystems have been made. Global "integrated assessment" models are now routinely used to evaluate impacts of climate change, including agriculture – for example, Nelson et al. (2014) summarizes and compares results from some of the major agricultural models. Also, there are national policy models, as well as many more detailed farm-level and landscape-scale models used to study impacts of technologies and policies on agricultural systems, including "climate smart" technologies (Antle and Valdivia 2021).

What We Know	What We Need to Know
Key sustainability indicators for agriculture and food system, including principal greenhouse gases (GHGs)	Current and future projection of GHGs and other sustainable development indicators for major production regions and systems and their uncertainties
Main sources of agricultural emissions	Actual and potential Climate Smart Agriculture (CSA) technologies for major production regions and systems
Technical potential and uncertainties of actual technologies to reduce emissions and offset of non-ag emissions	Assess CSA contriubutions and uncertainties towards NNE goal, and implications for sustainable development, under plausible future socio-economic and policy scenarios
Bio-physical data and models for crop yields, soil C and N2O, forest growth	Design protocols for bio-physical model ensembles, assess model uncertainties, improve data and models, improve usability for policy analysis and implementation
National economic policy models	Design protocols for national economic model ensembles, assess model uncertainties, improve models for climate policy analysis
Regional land use, technology impact assessment models	Design protocols for regional land use and technology impact assessment model ensembles, assess model uncertainties, improve models for climate policy analysis, develop methods for linkage to bio-physical and national economic models

Table 1. Towards a Science and Policy Consensus for Agriculture's Contribution to NNE

Technical and Economic Feasibility of Agricultural Greenhouse Gas Mitigation

Other chapters in this report document the technical options now available to reduce net greenhouse gas emissions from crop and livestock agriculture. Earlier studies focused on technologies available, by exploiting effects of changes in soil management to store previously lost carbon from cultivated soils, as well as potential for afforestation to store carbon in trees and other forms of above-ground biomass as well as soil, as well as the use of biofuels (Paustian et al. 2006). Studies showed that, using technologies available at that time, such as adoption of no-till grain production, and afforestation of suitable lands, carbon could be accumulated and stored in soils and above-ground biomass at a cost competitive with other forms of emission reductions. The amount of carbon that could be sequestered by agriculture was found to vary across regions, and depend on each region's environmental conditions, the feasible land use and management options in each region, the socio-economic characteristics of farms, and the economic incentives provided to farmers to sequester carbon.

A key insight from the early research was to distinguish the *technical potential* for greenhouse gas mitigation – i.e., the amount that was scientifically and technologically possible – from the *economic potential* – i.e., the amount that farmers would do given their economic circumstances, capabilities, and motivations. Research showed that there could be a large gap between the technical and economic potential for greenhouse gas mitigation, analogous to the *yield gap* discussed in Chapter 4 of this volume.

A lesson from early research is that there are a relatively small number of critical values or "parameters" in understanding the economic potential for agricultural greenhouse gas mitigation. One key technical parameter is the amount of change in emissions that a change in management practice can produce on, say, a given unit of land; a key economic parameter is the cost to the farmer of making that change. The ratio of these two parameters gives a fundamental piece of information – how much mitigation can be achieved per dollar of effort. Another key piece of information is how much financial reward per unit of emissions reduction – if any – the farmer receives for this change. Clearly, the greater the reward compared to the cost per unit of emissions, the more likely the farmer will be to make the change.

Another critical insight is that, across the millions of acres of cropland, rangeland and forest land in the United States, these key parameters vary widely. Thus, we can conclude that whatever the technical potential may be, the economic potential for farmers to actually make management changes that are "climate smart" will depend on key economic factors and how these factors vary across the landscape and over time. In additional to these technical and economic factors, research also has shown that farmers' willingness to make "climate smart" management changes will be impacted by their attitudes toward risk, political and social attitudes and settings, and other "behavioral" considerations (Zilberman and Pannell 2020).

Additionality, Permanence and Slippage

We noted above that in the 1990s, there was much resistance to including agricultural activities as part of proposed climate policies, and we see similar objections being raised in current public debate over the proposals for agriculture. The early literature identified several factors that could limit the usefulness of agricultural and forest mitigation efforts. One is the idea of ***additionality***. The goal of climate policy should be ***additional*** reductions in greenhouse gases above and beyond any that farmers or forest owners would do absent the climate policy. Combined with the fact that agricultural processes play out over varying periods of time (soil C can change fast or slow; forests grow slowly but can burn rapidly), this raises several challenging issues for policy design. For example, in the current policy discussions, some farmers who have already adopted conservation practices that add and maintain carbon in soils argue they should be paid for that carbon. Such payments may be viewed by some as equitable and might be politically expedient to garner support for a policy, but they raise the cost of achieving additional emissions reductions. Furthermore, poorly designed policies could lead to perverse incentives, e.g., farmers who anticipate future financial rewards for adopting climate smart practices may delay adoption today or undo previous climate smart practices.

Another issue in the design of policies is known as ***permanence.*** The goal of climate policy is to reduce greenhouse gas concentrations in the atmosphere permanently. Yet, some agricultural management changes, notably the "sequestration" of carbon in soils through photosynthesis and root growth, and incorporation of organic matter into the soil, can be reversed through disruption of the soil by tillage. Carbon captured in trees can be released by wildfires or use of wood for fuel. Moreover, designing "contracts" for farmers or forest owners to permanently keep carbon stored in soils or biomass can be complicated by leasing arrangements and other land ownership issues as well as magnitude of the opportunity costs to maintain the carbon credits in perpetuity. Further discussions of the concerns regarding permanence can be found in Willey and Chameides, 2007, Fei and McCarl 2023.

A third complication is the issue of ***leakage or slippage.*** For example, a policy in one country of encouraging planting and maintaining trees and not harvesting can raise the price of wood elsewhere and cause more harvesting, thus offsetting the global benefits of the policy. Likewise, conservation policies in a large country like the United States – for example, the Conservation Reserve Program that takes millions of acres of land out of production – can lead to more land being cultivated elsewhere, releasing carbon stored in soils. Given the global nature of agricultural markets, these issues raise substantial challenges to local or national policy effectiveness. Examples of research and challenges on slippage in forestry include Wear and Murray (2004) and Murray et al (2004)

Closely related to the above issues is how changes in greenhouse gas emissions can be quantified and the emission reductions tracked. For example, if farmers participate in a carbon market by supplying carbon offsets based on soil carbon sequestration or afforestation, a system would be needed to quantify and verify the amount of carbon being stored. The cost and reliability of doing this has been widely debated – clearly, a scientific consensus is needed on these technical and transactional issues to move forward with actual policies (Antle et al. 2003).

Policy Design Options

The other chapters in this volume show that agriculture has a technical potential to reduce its own emissions and offset emissions in other sectors by making changes in the ways that crops and livestock are produced. The history of U.S. agriculture has demonstrated that ongoing advances in agricultural productivity have come, and likely will continue to come, from ongoing investment by both the public research institutions in government and universities, particularly for the more basic science advances that are "public goods," as well as private sector research investment in the application of basic science to develop new technologies. Development of new technologies is only part of the answer – farmers must use (adopt) these technologies. And in the case of climate smart technologies, there are many unanswered questions. Will these technologies have attributes that make them attractive alternatives to the "conventional" technologies now in use? And if not, what will be needed to motivate farmers to use them? Are these motivations generated from the demand side of the market or through supply side incentives, or both? This is the challenge of policy design, and there has been relatively little investment in the economic research needed to address the adoption of currently available climate-smart technologies. To begin to be successful with the NNE goals, there will need to be sustained economic research to evaluate the potential adoption and impacts of new technologies as they are developed.

The first step to understanding the importance of climate policy is to observe that action by consumers or farmers without additional climate policy support is not likely to realize the potential for agriculture to contribute to achieving the NNE goal. On the "supply side," the reason is because reducing emissions in most cases is costly to farmers. A good example is the use of "precision management" technology, such as variable rate seed and fertilizer applications enabled with machinery equipped with global positioning system technology. Adoption varies across the United States and research shows that it is not necessarily economically superior to conventional "uniform" management (Basso and Antle 2020).

Likewise, there are economic factors limiting the role that consumers can play in addressing the global climate problem. Certainly, there is a positive role for consumers to choose environmentally friendly products, but there is often a disincentive for consumers because "green" products often cost more than conventional products. Second, it is often difficult for consumers to convey their demand for green products through supply chains. Attributing to this complexity is defining what constitutes a green labeling and the logistics and cost of measuring and certifying product quality, especially when the quality involves elements of the production system. For example, it is difficult for the wheat market to distinguish wheat produced with no-till cultivation from wheat produced with conventional tillage.

Due to these limitations to voluntary solutions, policy is needed to drive the transition towards more sustainable, climate-smart agricultural and food systems. There are two broad classes of policy available to achieve emissions reductions: direct regulation of economic activities, and incentive-based policies. Direct regulation is sometimes most efficient and effective, but this is typically in cases such as reducing lead in gasoline and paint where the most effective policy action is to ban its use. In most economically valuable activities, however, a complete ban is not desired nor economically justified; and in these sectors direct regulation is very costly because conditions vary from farm to farm. This is particularly true for agriculture where there are millions of farms of varying types, sizes, and locations.

The introduction of the cap-and-trade system for reduction of air pollution in the 1990 Clean Air Act showed the power and efficiency of incentive-based policies to achieve a desired reduction in pollution. The key idea is for government to determine the maximum total pollution that should be allowed and allocate a corresponding number of tradable "emissions allowances" to polluting firms. Thus, firms that face a high cost of reducing their emissions can buy allowances from firms that can reduce their emissions at a lower cost. Additionally, the cost of allowances creates an incentive for firms to find production methods that are less polluting or lower-cost ways to reduce emissions. However, there are downsides to a cap-and-trade policy – notably the costs and complexities associated with administering and managing the market for tradable allowances, including how to measure emissions and ensure compliance.

The other incentive-based mechanism to reduce emissions of a pollutant is to tax the polluting activity. In the case of greenhouse gas emissions, a "carbon tax" (or more generally, a tax on carbon or other greenhouse gas content of fuels and other basic commodities) has been proposed and widely debated. The obvious advantages of a carbon tax are that it could be relatively easy to implement by taxing fossil fuels and other basic commodities at their source, generates tax revenues that can be used to offset the regressive nature of the tax and would create an economy-wide incentive to substitute away from fossil fuels to lower-carbon fuels. Moreover, this incentive would operate through both demand side (consumption) and supply side (production) of energy.

Equity is an important issue with all policy mechanisms, whether regulatory or incentive-based. Most environmental policies impact people differently depending on their incomes, consumption patterns, and other factors. For example, both a cap-and-trade policy and a tax on emissions would increase fuel costs, at least in the near term before substitutes such as electric vehicles became widely available at a competitive cost. People who spend a relatively large share of income on fuel would be more impacted. Likewise, farm producers using relatively energy-intensive practices would see their costs increase more than others, and could be put at a competitive disadvantage if they are selling their products in international markets and their competitors do not face similar policies.

What We Need to Know: Towards a Scientific Consensus for Agriculture's Role in NNE

Research has shown that agriculture can reduce its own emissions and may have the potential to offset emissions from other sectors of the economy. We have established that there are existing and potential technology and policy options to do this, most likely some combination of regulatory and incentive-based mechanisms. These options will involve inevitable tradeoffs among economic, environmental, and social outcomes and the challenge is how to evaluate the many options and tradeoffs, and to map out feasible pathways for agriculture to contribute to NNE. For example, currently we do not know which agricultural sectors can realistically achieve NNE, or whether agriculture overall can achieve net negative emissions. The question becomes more complex as we broaden the scope to include the food system components such as transport, processing, and distribution. Given the connections with other social and environmental goals, we need to understand how changes in agricultural systems to be "climate-smart" will impact other goals such as protecting water quality and ensuring food security. *What we can say is that answering these complex questions will require a systematic, coordinated research program that integrates the available information and enables the evaluation of a range of plausible future "pathways" for agriculture and the food system.*

The type of forward-looking analysis needed for U.S. agriculture has been pioneered by the climate science community to improve climate projections using large, global climate simulation models. A key element in this approach to science is the development and inter-comparison of multiple simulation models. Within the agriculture science community, this type of approach has been pioneered by the Agricultural Model Inter-comparison and Improvement Project (AgMIP; Rosenzweig et al. 2013; 2018). This kind of systematic, coordinated approach is now needed to establish a scientific consensus around the contribution that agriculture can make to the NNE goal.

Designing and Evaluating Sustainable Pathways to Net Zero

It is evident that given the current high dependence on fossil fuel energy, moving agriculture and the food system towards the NNE goal will require substantial change in the entire U.S. economy including agriculture and the food system. Moreover, these changes will inevitably involve tradeoffs among groups in society – from farmers and agribusiness to every food consuming household in society. The evidence-based scientific consensus we are calling for in this chapter is only possible if it comes from a participatory process that reflects the wide array of interests in society.

A major innovation in both industry and science is the development of new ways to envision plausible future pathways for the economy and society. These "foresight" methods are being used to project future climate changes and their impacts at global, national, and sub-national scales (Zurek et al., 2021). At the agricultural system level, participatory approaches are now being widely used to develop and assess agricultural technologies and development pathways (Valdivia et al., 2015). The development of future pathways begins with a narrative description of a future state of the world, followed by identification of key features of future systems in bio-physical, technology, economic and social dimensions, as well as identification of key sustainability indicators to be used for evaluation of pathways. Goals can be identified as key indicators, either by the stakeholder group, or by linking the pathway concepts to goals established by governmental processes.

Evaluating Agricultural Development Pathways

Most evaluations of agricultural system sustainability, as well as climate impact assessments, are implemented in a framework which integrates climate, crop, livestock, and economic data and models. Large-scale global assessments are implemented using components linked as illustrated in Figure 1. Future climate simulations are the first component of the assessment framework and are based on assumed trajectories of greenhouse gas emissions that are consistent with a plausible range of future socio-economic conditions. For the agricultural and economic model components, additional socio-economic pathway elements are needed, such as rates of population growth and rates of technological change. The agricultural and economic models project production, consumption, prices, and related outcomes at various spatial and temporal scales, depending on the type of model and objectives of the analysis.

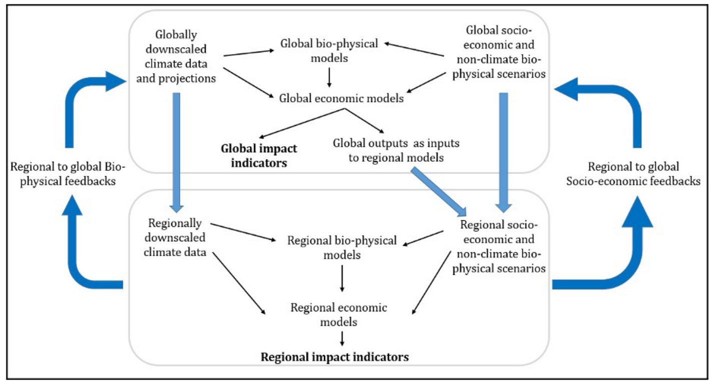

Figure 1: AgMIP Global and Regional Integrated Assessment Framework
Source: Antle and Ray (2020).

This type of integrated assessment method is being used at scales ranging from global to regional (i.e., multi-national or national), and sub-national (i.e., the U.S. corn belt). For example, Nelson et al., (2013) present from ten global models that are being used for climate impact assessments. The U.S. Department of Agriculture's Economic Research Service has developed a regional economic impact assessment model that is linked to a bio-physical model that simulates crop yields and other environmental outcomes for agro-ecological regions of the United States (Johansson et al. 2007). Van Ittersum et al., (2008) describe an integrated assessment framework created for the European Union. This framework links field-level bio-physical models with farm-level economic optimization models and a multi-country econometric policy model and is being used for policy analysis and climate impact assessment. van Wijk et al., (2014) and Kanter et al., (2016) reviewed a large number of studies that combine various types of bio-physical simulation models with economic optimization or simulation models to study the sustainability of agricultural systems and identify the strengths and limitations of the currently available models. Brown et al. (2015) reviewed the literature in the context of the U.S. food system.

Innovations in data, models, and methods

The diversity and heterogeneity of agricultural systems create many challenges for the agricultural system modeling portrayed in Figure 1. Most notably, these characteristics mean that large amounts of highly detailed, site- and time-specific data are needed. Thus, a key limitation to the development and use of agricultural systems models is data (Antle et al. 2017). Better data are needed to further improve crop and livestock models in ways that are useful for both on-farm management decision making and for use in research to develop and test new technologies, and to evaluate their productivity and sustainability. Fortunately, new technologies, such as the use of mobile sensors and other "big data" technology, are helping to bridge this gap, such as the technologies enabling digital agronomy and precision agriculture.

However, several important challenges must be overcome to make new digital data useful for both farmers and scientists (Capalbo et al., 2017; Antle 2019). One issue is how to share individual data while maintaining farmers' privacy and property rights to their data. Another issue is how to translate individual data, typically acquired using various non-standard, proprietary formats, into a generic format that would be FAIR (findable, accessible, interoperable, and reusable). Ideally, an integrated private-public data infrastructure that meets both private and public needs would make sense, but we are far from such a system today. Private data and related soft and hard infrastructure are being developed by a growing array of management advisory and technology companies. Data generated by individual producers or by private firms selling data or advisory services are not public and thus not findable or accessible, often even by farmers themselves. There are no established data standards being used, and thus data are not interoperable even when findable and accessible.

In addition to better data, new investments in analytical capabilities are needed to simulate the performance of future agricultural systems under projected future climates. There are substantial limitations to the models now being used to project crop yields under future climates (Jones et al., 2017). Likewise, economic models for evaluating new agricultural technologies at farm, regional, national, and global scales need improvements, and methods for their integration with bio-physical models and across scales need to be improved (Antle 2019; Antle and Valdivia 2021). A key issue with all computer simulation models is the uncertainty associated with their projections. The climate science community has established the use of protocol-based model inter-comparisons as a effective way to improve models, and the use of multiple model "ensembles" provides more reliable projections and ways to characterize model uncertainty. AgMIP researchers have demonstrated similar benefits of protocol-based inter-comparisons for agricultural system model improvement, and the value of multiple model projections (Rosenzweig et al., 2018). A major advance that is needed in this field of science is to develop the methods and computational methods to enable multi-model ensembles of models linked to simulate large complex systems such as agriculture. Other needed advances include the development of more open-source models and information technology tools for their application over large regions and globally (Janssen et al., 2017).

Towards NNE: Agriculture's Role in Current Policy Initiatives and Research Needs

Agriculture's role in NNE is direct and trackable. The most straightforward pathway to reduce agricultural greenhouse gas emissions would be to utilize the various existing USDA conservation, insurance, and subsidy programs to incentivize adoption of climate smart agricultural practices. The idea of a government-supported "carbon bank" has been proposed by the Biden Administration. However, USDA currently lacks the capability to systematically evaluate the contribution that current or proposed future policies (say, in the next Farm Bill) would make towards the NNE goal. Past investments have established many useful tools – for example, the COMET tool developed by the National Resources Conservation Service for farmers to evaluate their own emissions. However, to provide credible estimates, for example for a carbon market, the uncertainties associated with the emission levels produced by this tool need further evaluation and refinement. Moreover, this tool is designed for individual farms. At the regional and national scale for policy analysis, a rigorous, multi-model approach is needed for major farming systems across the diverse regions of the United States that can be aggregated to achieve a national benchmark and evaluation. Given the inherent uncertainties in climate projections, it is essential that this framework be based on established protocols to facilitate multi-model analysis, such as those being developed by the Agricultural Model Intercomparison and Improvement Project (Rosenzweig et al., 2018).

The need for a science base and verification system for emission reductions for agricultural climate policies is also evidenced by recent private sector initiatives to offset emissions in other sectors of the economy. One example is Microsoft Corporation's hesitancy to buy carbon credits from farmers due to perceived uncertainties and lack of standards (Reuters New Service 2021). To address this need, Microsoft is also working with agricultural firms such as Land O'Lakes to develop databases to track production practices, farm health and environmental conditions and develop analytical tools to support a carbon market for agriculture (Agfunder Network 2021). Private-public partnerships are essential to bring together science and industry to build these much-needed new capabilities.

In summary, *the key research opportunity we have identified in this chapter is to build a research program to design an effective climate policy for agriculture that will support the national NNE goal*. The current debate over national policy proposals illustrates the value that this type of research program would provide. Four overarching recommendations to build the research program are noted in the introduction with more specific details provided throughout the chapter and highlighted below:

- Extend the national-level assessments of *climate impact* to a framework for the evaluation of agriculture's potential contribution to the NNE goal and use this new evaluation framework to assess the effects of policies proposed for agricultural producers within the larger context of broader national climate policies.
- Distinguish technical potential for greenhouse gas emission reductions from the economic potential and design and fine-tune policies based on a clear understanding of location and farmer-specific economic factors that will impact adoption.
- Create tracking systems or tools for quantifying changes in greenhouse gas emissions that accounts for additionality, permanence and slippage among various mitigation and emission reduction policies.
- Scale up investments in data and analytical tools to evaluate the potential impacts of existing and prospective technologies can make toward reducing emissions.
- Create opportunities to partner with industry and public sector to ground truth new technologies and design effective tracking systems for emission reductions and carbon credits.
- Quantify adoption rates of climate-smart practices under alternative agricultural policy options and how these rates are impacted by changes in social, economic and ecological conditions.

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Chapter 9: Summary of Results and Priority Research Needs

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Introduction

This report analyzed the potential of U.S. agricultural production supply chains to sequester carbon and reduce greenhouse gas emissions. The approach was to identify the state of knowledge on the benefits and possible implementation of specific practices necessary to create negative GHG emissions in agriculture and to create a roadmap for implementing the most promising strategies to achieve greenhouse gas negative agriculture. The total U.S. agriculture GHG emissions after land sequestration in 2020 was approximately 595 Tg CO_2e (approximately 0.6 Gt) (USEPA, 2021). To achieve GHG negative agriculture in the U.S., cumulative emissions must be reduced by at least 0.6 Gt CO_2e per year.

The theory behind reducing net greenhouse gas emissions from agricultural production across the U.S. is that changing technologies and practices implemented by producers will lead to lower greenhouse gas emissions and more carbon sequestration in the soil. The preceding chapters identified technologies and practices that demonstrate the highest reductions of greenhouse gas emissions and the highest potential for implementation. The potential for changing the agricultural carbon budget was categorized in terms of closing the practice gap and accelerating the implementation of new technologies, often referred to as regenerative agriculture practices (see Chapter 2). Closing the gap for each metric was estimated at two levels: Medium and High adoption rates. Medium adoption rates across each metric assumed the adoption gaps were reduced by 50 percent, and high adoption rates assumed the gaps were reduced by 75 percent. Research priorities were defined as those knowledge gaps that must be filled to address high priority practice implementation and high priority technology development. High priority practices and technologies were determined through materiality matrices analysis of the potential for effects and timelines for impacts. This assessment attempted to highlight the most effective practices and technologies, but other innovations and opportunities are beyond those considered here.

Soil Carbon Sequestration

The six principles of regenerative agriculture described in Chapter 2 include 1) minimize soil disturbance, 2) maximize production diversity, 3) keep the soil covered, 4) maintain living roots, 5) integrate animals, and 6) understand the local agroecosystem context. The practices that drive these six principles include conservation tillage (especially no-till), deep tillage of soils with high SOC in the upper layer, crop rotation, bio-char additions, leaving crop residue on the surface and planting cover crops, livestock grazing, and regenerative agriculture practices that increase nitrogen use efficiency.

With aggressive (high) adoption rates across U.S. agriculture, soils could capture 234 Tg $CO_2e\ yr^{-1}$, an increase of 1.47 Mg $CO_2e\ ha^{-1}\ yr^{-1}$ over existing practices. Moderate (medium) levels of adoption would result in increased soil sequestration of approximately 133 Tg $CO_2e\ yr^{-1}$ (Table 9-1). The importance of optimized strategies based on the control functions for increasing SOC cannot be overstated. Soils in humid climates or irrigated systems can sequester almost three times the mass of C than in drier climates (2-3 Mg $CO_2e\ ha^{-1}\ yr^{-1}$ compared to < 1 Mg $CO_2e\ ha^{-1}\ yr^{-1}$). Degraded land also has a higher sequestration capacity than undegraded land because the soil organic carbon has been depleted.

Managing and documenting the levels of carbon sequestration in soil will require developing novel technologies to measure soil carbon, methods that can economically assess soil parameters at sub-field (100 m^2 or 0.01 ha) scales. Such monitoring technologies do not exist. Making decisions based on these data will require ecoregional and crop specific decision support systems based on more sophisticated models than are currently available. Translating model output to decision support information will require developing risk-based assessment methods useful to producers in real time.

Nitrogen Use Efficiency

Nitrogen fertilizer is the most energy intensive and GHG emitting component of modern agriculture, as described in Chapter 3. Improving nitrogen use efficiency will require reducing N loss as NH4 and N_2O to the atmosphere and NO3 and NH4 loss to the hydrosphere. Nitrous oxide emissions contribute almost 57 percent of all agricultural GHG emissions on a CO2-equivalent basis (USEPA, 2021). Agricultural soil management practices (fertilization and tillage) drive 94 percent, or 316 Tg CO_2e of U.S. N_2O emissions. Reducing N_2O emissions will require integrated, subfield precision management of nitrogen fertilization. Precision conservation farming could reduce or eliminate tillage in low productivity, and thus sub-profitable, parts of fields resulting in no N fertilizer application for those areas. High adoption rates of precision conservation could result in a cumulative reduction of GHG of 70 Tg CO_2e of U.S. N_2O emissions. Medium adoption rates could result in a cumulative reduction of GHG of 40 Tg CO_2e of U.S. N_2O emissions.

Spatially variable N application rates matched to crop uptake requirements will further reduce GHG effects from N fertilization. Potential N_2O emissions reductions with high and medium adoption of spatially variable N application methods described in Chapter 3 are 114 and 75.2 Tg CO_2e yr^{-1}. Increased nitrogen use efficiency will require high resolution (100 or 0.01 ha) yield maps, in-season remote sensing imagery, weather forecast, and soil mineralization potentials from crop and soil simulation models (Basso et al., 2019).

Reducing on-farm energy use and crop yield gap is closely tied to N_2O emissions. Economic monitoring technologies do not exist to measure soil characteristics at this scale. As with SOC, making decisions based on these data will require ecoregional and crop specific decision support systems based on more sophisticated models than are currently available. Translating model output to decision support information will require an additional level of risk-based assessment currently not available.

Direct Energy Use Reductions

Direct energy use includes primary fuels used in motorized activities and energy used to power pumps, lights, fans, and other on-farm equipment (Chapter 6). Diesel and electricity consumption represented 55 and 36 percent of direct energy use GHG emissions on farms. The next highest impact was from natural gas at less than 5 percent. Focusing on reducing diesel and electricity consumption at the farm level could reduce farm energy use the most and thus have the largest effect on GHG emissions. Indirect energy emissions were predominantly from nitrogen fertilizer production and were addressed in Chapter 3. Diesel consumption could be offset by producing on-farm cellulosic biofuels grown on marginal lands. High and medium adoption rates could reduce petrochemical diesel consumption on farms resulting in GHG emission reductions of 38.9 and 25.7 Tg CO_2e yr^{-1} CO_2e . Similarly, generating electricity on-farm using solar and wind technologies could reduce reliance on grid-distributed electricity by 75 percent with high adoption rates. High and medium GHG emissions reductions from on-farm solar and wind electricity generation were estimated at 25.4 and 16.8 Tg CO_2e yr^{-1}.

Row Crop Yield Gap

Crop yield gaps represent losses of potential production for each unit of input (energy, water, land, etc.) and each impact (GHG emissions, water pollution, soil erosion, etc.) (Chapter 2). Achieving potential yield for row crops is very high if operators use improved genetics combined with optimized practices like legume crop rotation, cover crops, integrated irrigation and fertilizer application, and others. The potential reduction in GHG emissions from these activities was provided in chapters 2-6. Chapter 2 details the research goals necessary to achieve these goals.

Animal Protein Production

Animal agriculture is responsible for 39 percent of the total agricultural GHG emissions in the U.S. (232 Tg CO_2e yr^{-1}) (Chapter 5). The primary sources of GHG emissions from animal agriculture are in crops fed to the animals and associated with CH_4 from enteric methanogenesis and N_2O and CH_4 emissions from manure management. Reducing emissions in crop production will also reduce feed impacts in animal production. To avoid double counting, those reductions are credited to crop production practices (chapters 3 and 4).

Feed use efficiency represents a significant opportunity to reduce GHG emissions across all animal production sectors. Combining improved digestibility, feed additives that improve nutrient uptake by the gut, and diet optimization could increase animal productivity (meat, eggs, milk) while reducing direct GHG emissions (enteric methanogenesis) and manure production per unit of product. Further improvements could be achieved through feed additives such as 2NOP and ration changes to reduce enteric methane. Feed use efficiency with current technologies could reduce GHG emissions by 25% across all animal production systems. Assuming high adoption rates would achieve an average of 75 percent effectiveness and medium adoption rates would achieve an average of 50 percent effectiveness, the high and medium GHG emissions reductions would be 43.5 and 25 Tg CO_2e yr^{-1}.

Beef production is responsible for 80 percent of animal agriculture GHG emissions, or seven percent of total agricultural emission reductions. Improved grazing management practices by rotating pasture could reduce GHG emissions by 0.4 Mg/ha-yr CO_2e . High adoption rate for this practice was estimated at 75 percent across the 214 million ha of U.S. grazing land and medium adoption rate was estimated at 50 percent of that area, resulting in potential GHG emissions reductions of 64.1 and 29.9 Tg CO_2e yr^{-1}.

Animal manure is responsible for as much as 20 percent of total agricultural GHG emissions (46.4 Tg CO_2e yr^{-1}). Manure management across all of agriculture could reduce manure GHG emissions by as much as 40 percent, especially if manure is integrated with soil when applied to the land. Manure management technologies include housing, waste collection, waste treatment, and land application. Assuming high adoption rates would achieve an average of 75 percent effectiveness and medium adoption rates would achieve an average of 50 percent effectiveness, the high and medium GHG emissions would be reduced by 14 and 9 Tg CO_2e yr^{-1}.

Food Loss and Waste

The challenge and extent of food loss and waste post-production is confounding (Chapter 7). Nearly 17 percent of crops are never harvested (roughly 16.7 million tons, mostly specialty crops), and 40 percent of all food produced in the U.S. is lost or wasted, with 80.6 million tons unsold or uneaten. This wasted food is directly responsible for 238 Tg CO_2e/yr, not including processing, distribution, preparation, and landfill and composting emissions. Assessing food waste reductions against GHG emissions in the U.S. will not necessarily involve reduced production of agricultural and food products. The benefits of reducing food waste and thus reducing GHG emissions will rely on changes in food demand from a growing and more prosperous population.

Risk Analysis of GHG Emissions Reduction Potential

The medium and high adoption rates for each GHG reduction practice were analyzed using a quantitative risk approach to calculate how well implementing existing practices could achieve net carbon negative agriculture in the U.S. The mean GHG reduction for medium and high adoption rates were estimated for 10 practices (Table 9-1). Potential minimum and maximum GHG emissions reductions were estimated using a quartile distribution for each mean impact from medium and high adoption rates, where the minimums were 75% of the mean, and the maximums were 125% of the mean. The minimum-mean-maximum ranges were assumed to represent triangular distributions of GHG reduction potential, with the mean as the central

Process	Practice	Medium Adoption			High Adoption		
		Min	Mean	Max	Min	Mean	Max
Soil C Sequestration	Corn ethanol to herbaceous biomass crops	17	23	29	6	23	29
	Perennial cropping systems on marginal lands	3	4	5	2	7	9
	Conservation tillage	80	106	133	51	204	255
Nitrogen Use Efficiency	Precision Conservation	30	40	50	18	70	88
	Spatially variable N application	56	75	94	29	114	143
Direct Energy Use	Cellulosic Biofuels Production	19	26	32	10	39	49
	On-farm Solar and Wind	13	17	21	6	25	32
Animal Agriculture	Feed Use Efficiency	19	25	31	11	44	54
	Improved Grazing Management	22	30	37	16	64	80
	Manure Management	7	9	11	4	14	18

value and minimum and maximum as the lower and upper boundaries (Table 9-1). Monte Carlo simulation modeling was performed with triangle distributions for each practice at medium and high adoption rates (Olea, 2011). Individual practices and cumulative carbon emission offsets were simulated with 5,000 random iterations to calculate the mean and 90 percent confidence intervals of impact for each practice adoption level.

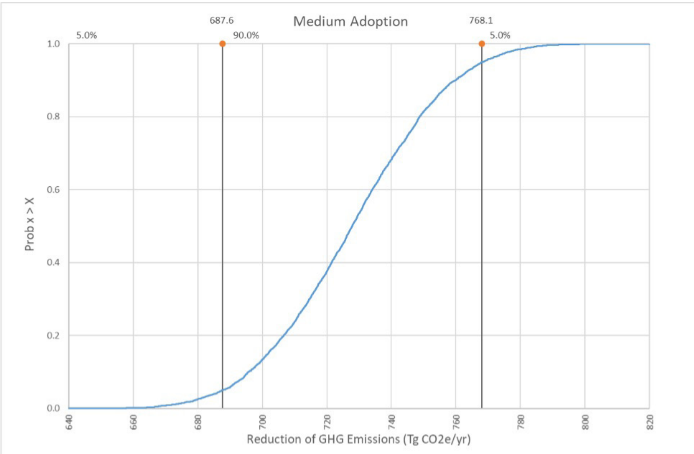

Figure 9-1

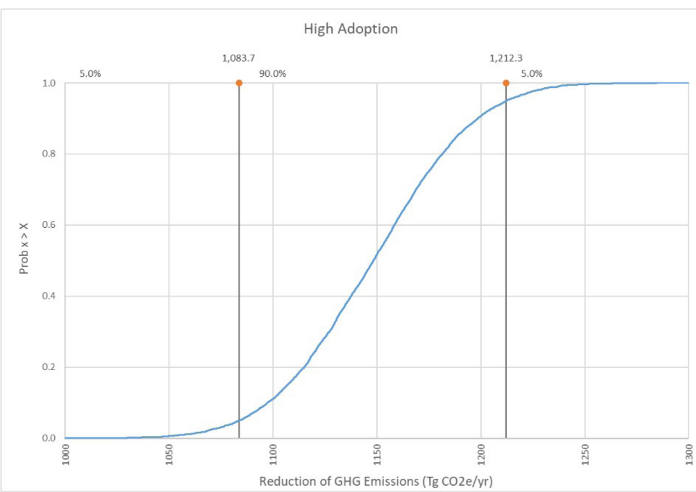

Figure 9-2

Medium adoption scenario for all 10 practices showed a mean offset of GHG emission of 355 Tg CO_2e /yr with a 90 percent confidence that offsets would be between 330 and 379 Tg CO_2e /yr (Figure 9-1). The high adoption scenario across all 10 practices showed a mean offset of GHG emission of 604 Tg CO_2e /yr with a 90 percent confidence of GHG reductions being between 560 and 648 Tg CO_2e /yr (Figure 9-2). These results suggest that high adoption rates of the 10 practices could offset more GHG emissions than current annual agricultural emissions at least 60 percent of the time.

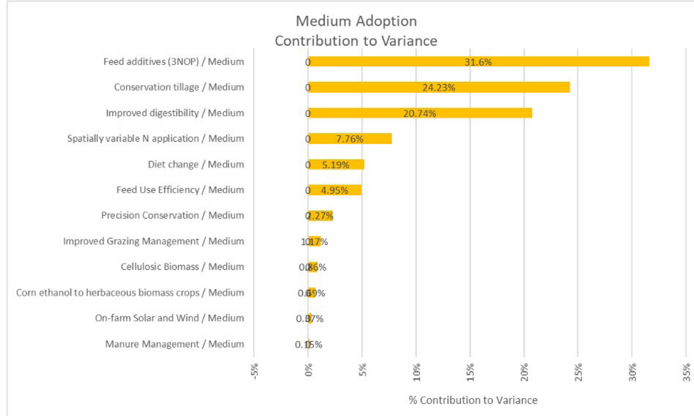

Figure 9-3

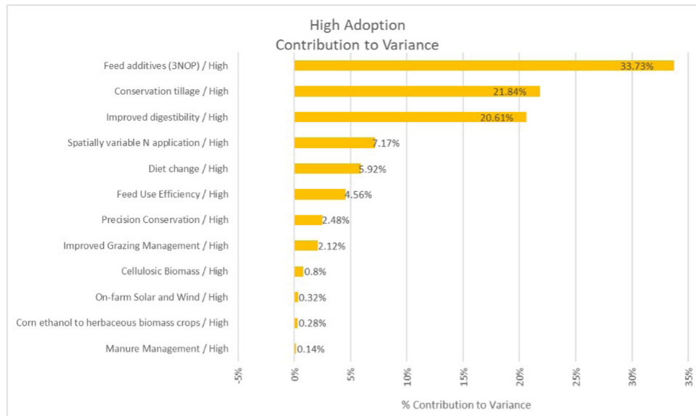

Figure 9-4

A sensitivity analysis for each scenario was performed (figures 9-3 and 9-4). Across both medium and high adoption rates, the three most effective practices were conservation tillage, spatially variable N application, and precision conservation, representing more than 40 percent of potential GHG emission reductions. The next four most effective practices were improved grazing management, cellulosic biofuel production, converting corn ethanol production to herbaceous biomass, and feed use efficiency.

Conclusions

This report aimed to analyze the potential of U.S. agricultural producers to sequester carbon and reduce emissions. The approach was to identify the state of knowledge of the benefits and practical constraints for adopting specific practices to create negative GHG emissions in agriculture and to create a roadmap for implementing the most promising strategies to achieve GHG-negative agriculture. This report evaluated the most effective practices to sequester soil carbon, use nitrogen efficiently, reduce on-farm energy use, optimize yield, use animal feed efficiently, optimize economic management, and reduce food waste. This is not a comprehensive list of all potential GHG reduction approaches but covers the best practices. Challenges of scale are apparent in this assessment. National data are, of course, highly variable and often unreliable. Global scale assessments are even more difficult for these same reasons. However, a national assessment of potential GHG emissions reduction strategies would be valuable in setting a benchmark of the possibilities by prioritizing implementation and innovation strategies.

While these analyses showed that current best carbon management practices, even if adopted across most U.S. agricultural lands, are unlikely to result in net carbon negative agriculture, high adoption rate of these practices current practices could, with high probability, move U.S. agriculture very close to net carbon neutral emissions. With the current U.S. agriculture emissions of 595 Tg CO_2e /yr, high adoption of the 10 best current practices would offset 110% of U.S. agricultural emissions. With further research and development, agriculture could surpass this 110% even with modest gains in technology and efficiency. Each best carbon management practice represents a portfolio of practices and technologies necessary for effective implementation. For example, conservation tillage holds the highest potential of reducing net emissions. The technologies that allow conservation tillage (including no-till practices) include genetic optimization of crops for specific production conditions; disease, weed, and insect control to reduce yield losses; and subfield management. More effective sensors for SOC, soil moisture, and other soil characteristics to support subfield management decisions are also critical. These are the same innovations necessary for improving NUE and WUE.

Research and Innovation Priorities

Based on this assessment, the research priorities necessary to achieve net greenhouse gas negative agriculture include but certainly are not limited to the following:

- Conservation tillage
- Improved animal feed digestibility
- Spatially variable N application
- Diet change for animal production
- Feed use efficiency
- Precision conservation
- Diverse crop rotations with cover crops
- Improved grazing management
- Cellulosic biomass production
- On-farm solar and wind power
- Corn ethanol to herbaceous biomass crops
- Manure management

The units of analysis in this report were Global Warming Potential, using the IPCC 100-year equivalency values. However, analyzing a sequestration strategy using GWP values is difficult due to the definition of GWP 100; soil carbon would need to remain sequestered for 100 years to achieve the GWP goals: the well-known problem of permanence. A more appropriate approach might be a mass balance for each category of gas (CO_2, CH_4, and N_2O). This approach would treat sequestered and emitted GHG as equivalent and may provide a more accurate accounting for a GHG budget and credit approach. However, this analysis aimed to evaluate the range of possibilities for agriculture to respond to GHG emissions reductions with the practices available today. The decision was made to move forward with GWP values as cumulative estimators.

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Potential for U.S. Agriculture to Be Greenhouse Gas Negative

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