Carbon Science for Carbon Markets: Emerging Opportunities in Iowa. Chapter 8. Agricultural Carbon Planning

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Carbon Science FOR Carbon Markets

Emerging Opportunities in Iowa

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OF SCIENCE AND TECHNOLOGY

Abstract

Credible carbon credits are a precondition for carbon markets. Unlike two decades ago, when voluntary carbon markets were just being developed, much is known today that supports credible carbon credits, including those that are agriculturally based. This report addresses ways to further improve the credibility of agricultural carbon credits and reduce the cost of carbon programs by assessing the underlying science and adding transparency to how carbon markets function. We assess the history and structure of carbon markets; carbon credit measurement, reporting, and verification protocols; the impacts of land—especially cropland—and livestock management practices on greenhouse gas and soil organic carbon dynamics, and adoption of these practices; existing and emerging engineering technologies that could reduce greenhouse gas emissions or enhance carbon removal; and quantitative tools that could help facilitate carbon market development. The geographic scope is primarily focused on the state of Iowa. The report furthermore highlights ways multisectoral collaborations—for example, between farmers, scientists, industry, government, and civil society organizations—could remove barriers and further market development.

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Contents

Chapter 1. Overview of Carbon Science for Carbon Markets: Emerging Opportunities in Iowa

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1.1 Background on Carbon Sequestration Task Force

The Carbon Sequestration Task Force was authorized on June 22, 2021, by Iowa Governor Kim Reynolds under Executive Order Number 9. Task A of this executive order states: “Reviewing the research on carbon sequestration, considering any gaps in current assessments, and determining whether new research, standards, or definitions should be developed.” The Task Force was given the following vision for their work:

> *Iowa will be the leading state for creating carbon value through agricultural stewardship and energy generation.*

As part of the analysis provided to the Task Force, researchers at Iowa State University (ISU) were asked by the Iowa Economic Development Authority (IEDA) to undertake an assessment of the science supporting agriculturally based carbon markets. This report includes the work of 51 faculty members, staff, postdoctoral associates, and graduate student scholars associated with four colleges, four institutes and centers, and 13 departments throughout ISU. This report demonstrates the extent of supporting science and expertise of its authors.

If there is one main takeaway from this report it might be this: while much is still unknown about carbon and greenhouse gas dynamics and the market potential of carbon credits, Iowa is advantageously suited to lead the nation in agricultural carbon management.

1.2 Key Terms and Concepts

Today’s carbon markets focus on carbon dioxide (CO_2) and other greenhouse gases (GHGs), including methane (CH_4) and nitrous oxide (N_2O), according to their carbon dioxide equivalent (CO_2e). All three of these GHGs trap heat in the Earth’s atmosphere and arise in part from human activities, including transportation, energy generation, industrial, residential, commercial, and agricultural activities (USEPA 2020). Carbon dioxide accounts for 80% of total human-generated emissions for the United States (USEPA 2021). As a result, reducing carbon dioxide emissions from burning carbon-rich fossil fuels has received most of the attention in efforts to mitigate climate change. Similarly, calls to remove GHGs already in the atmosphere have focused on carbon dioxide removal because it is not practical to remove other GHGs. This focus on emissions and accumulations of carbon dioxide arising from the burning of carbon-rich fossil fuels accounts for “carbon” serving as an imprecise proxy for all greenhouse gases in popular and scientific literature. While imprecise, “carbon” is likely to continue garner the focus of efforts to reduce atmospheric GHG levels, and is the focus of the growing voluntary market. Key terms crucial to understanding carbon markets include:

- **Carbon Credit** – Unit certified by a carbon credit program or standard that can be traded in carbon markets, representing one metric ton of carbon dioxide equivalent.
- **Carbon Dioxide Equivalent (CO_2e)** – The amount of carbon dioxide emission that would cause the same integrated radiative forcing or temperature change, over a given time horizon, as an emitted amount of a GHG or a mixture of GHGs (IPCC 2018).
- **Carbon Market** – A market in which a supply of carbon offset credits is sold to companies that use them to meet their voluntary or regulatory GHG emissions goals or requirements.
- **Carbon Offset** – Reduction, avoidance, or sequestration of one metric ton of carbon dioxide or greenhouse gas equivalent.
- **Carbon Removal** – The removal of carbon dioxide from the atmosphere through ecosystem processes or engineering technologies resulting in a net decrease of GHGs in the atmosphere. "Carbon dioxide removal" may also be used.
- **Carbon Sequestration** – The process of storing carbon in a carbon pool (IPCC 2018). Typically refers to sequestration of atmospheric carbon dioxide. Can be through biophysical or engineering processes.
- **Carbon Sink** – A reservoir (natural or human-made, in soil, ocean, and plants) where carbon dioxide is stored (IPCC 2018).
- **Carbon Storage** – The process of storing carbon in a pool. "Carbon sequestration" is often used to mean "carbon storage," but not always synonymously.
- **Compliance Market** – A market for carbon offsets where entities emitting GHGs above a certain threshold are required by law to cut emissions and/or purchase credits. This typically applies to airlines, industries, and power generators.
- **Voluntary Market** – A market for carbon offsets where entities opt to cut emissions and/or purchase credits but are not required to do so by regulation. Examples include corporations and governments who purchase credits based upon a policy goal.

Some of these terms are used in multiple ways and many have overlapping meaning; for example, carbon sequestration and carbon storage. Distinguishing between emissions associated with carbon that was sequestered for exceptionally long periods of time, such as associated with the burning of fossil fuels, and those associated with shorter-term natural processes occurring in the biosphere is crucial to developing strategies for reducing atmospheric GHGs. It is thus important to understand the key concepts of "carbon positive," "carbon neutral," and "carbon negative" emissions.

Carbon Positive – The flow of GHGs into and out of the atmosphere can be compared to filling and emptying a swimming pool with water, as shown in Figure 1.1. GHG emissions into the atmosphere, denoted as "carbon positive" originate from carbon that had been sequestered for tens of thousands to millions of years. Its entry into the atmosphere can be likened to water flowing from a storage tank into a swimming pool. In this case, the rise of GHG concentration in the atmosphere is analogous to the water level in the swimming pool rising: we can turn off the flow of water from the storage tank to the pool to stabilize the water level, but this does nothing to reduce the amount of water in the pool.

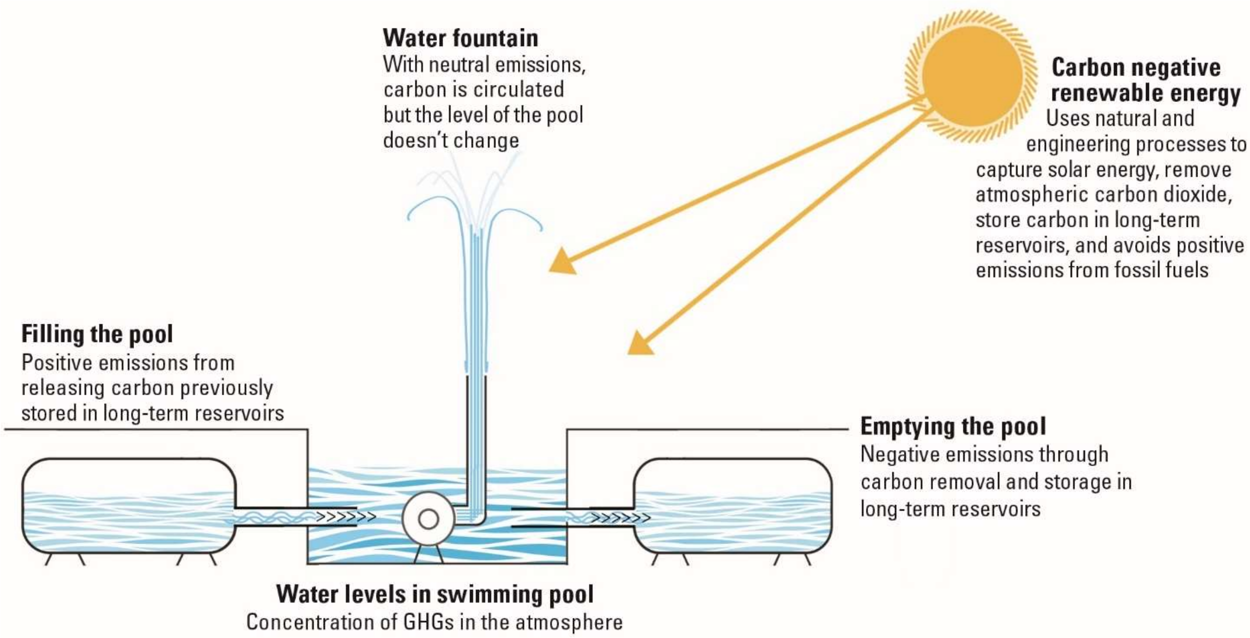

Figure 1.1. Understanding carbon flows into and out of the atmosphere through an analogy with water flowing into and out of a swimming pool.

Carbon Neutral – "Carbon neutral" emission occur when human-caused emissions of GHGs to the atmosphere are balanced by human-caused removals over a specified period. Again, using the swimming pool analogy, the fountain shown at the center of Figure 1.1 circulates water between the pool and the surroundings without changing the water level in the pool. This is analogous to renewable fuels or materials from biomass, which circulate carbon dioxide between the atmosphere and the biosphere and in the process provides carbon neutral energy and products for society. While energy produced from fossil sources and biomass both emit carbon dioxide into the atmosphere, the net flow of carbon dioxide is positive for fossil fuels, but can be neutral for biomass energy if proper production practices are used.

Carbon Negative – The term "carbon negative" is applied to human processes or products when associated carbon dioxide removal from the atmosphere exceeds human-caused carbon dioxide emissions. Just like lowering the water level in the swimming pool by pumping water into a second storage tank (Fig. 1.1), the concentration of carbon dioxide in the atmosphere can be lowered through ecological or engineering processes. Renewable energy avoids positive carbon emissions from fossil fuels, while carbon neutral and carbon negative emissions require carbon dioxide to be removed from the atmosphere. Nature uses photosynthesis to capture solar energy, and in the process removes atmospheric carbon dioxide by converting it into biomass. By combining multiple strategies—including harnessing photosynthesis, using renewable energy sources, reducing overall GHG emissions, and contributing to pathways for storing carbon in long-term reservoirs—agriculture has the potential to become carbon negative and contribute to climate change mitigation.

In this report, we attempt to be clear and unambiguous in our use of language. A comprehensive glossary of key terms is provided in **Appendix 11.1**.

1.3 What is the Market Potential?

Emerging carbon markets necessarily involve agriculture because of agriculture's contribution to GHG emissions (Fig. 1.2) and because photosynthesis by green plants is the most effective way to rapidly remove carbon dioxide from the atmosphere. Iowa has a unique competitive advantage for engaging in emerging carbon markets due to the extensive scale and high productivity of agriculture within the state. Carbon markets further offer farmers a way to pay for practices that help sustain crop yields and provide additional ecosystem services in the long term.

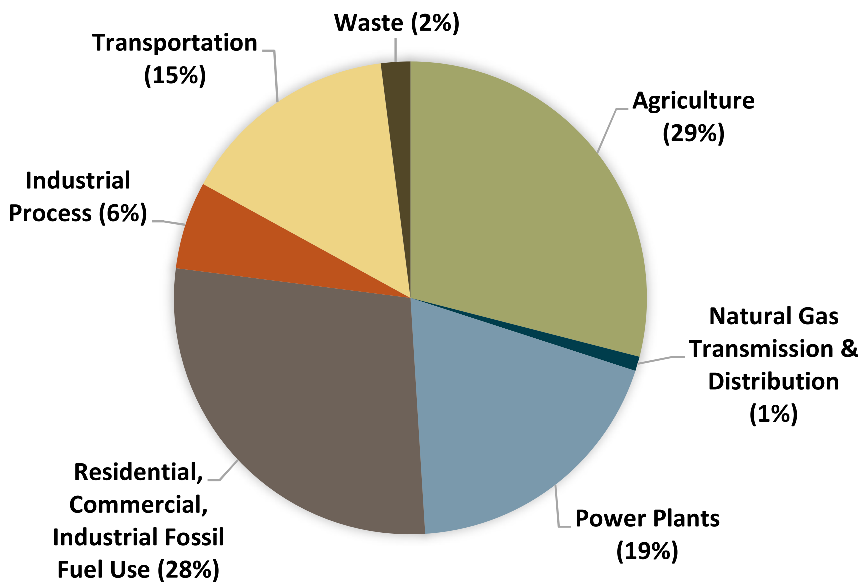

Figure 1.2. Iowa greenhouse emissions by sector (IDNR 2020).

Carbon buyers are willing to pay for both reductions in GHG emissions and carbon removal. At present, however, there are gaps in the research on the effectiveness of specific agricultural practices and technologies to meet these goals. While there is growing knowledge of how much carbon is being produced in various sectors, there is less understanding of the amount of carbon currently stored in the soil, the capacity for further storage, and the extent to which it can be retained moving forward. A comprehensive, science-based assessment is needed to quantify the impacts of soil management, land use, edge-of-field practices, and engineering technologies. Transparent and credible assessment allows farmers, Iowa agriculture, and carbon credit buyers to better understand and receive the full benefits from emerging carbon markets over the long term and, importantly, lower the costs of measuring, reporting, and verification.

With the currently existing technologies, photosynthesis in plants is the most effective and adaptable method of removing excess carbon dioxide from Earth's atmosphere. The highest levels of global plant productivity are measured in the United States Corn Belt due to the convergence of natural and human factors (Fig. 1.3; Guanter et al. 2014). In addition to peak rates, high rates of photosynthesis occur over a large area in the Corn Belt, meaning that the region can provide carbon removal at a scale that is efficient for businesses to offer carbon programs and attractive to investors in carbon removal or related technologies.

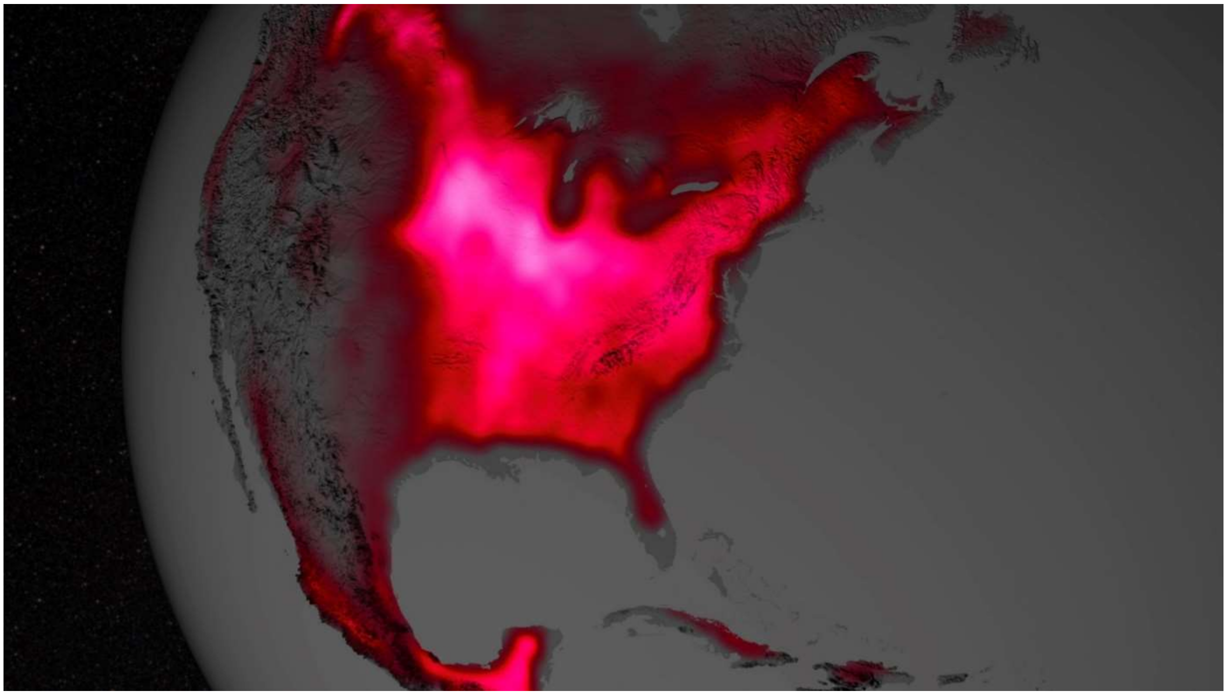

Figure 1.3. Sun-induced photosynthetic florescence, a measure of gross primary production, for North America during the northern hemisphere summer, with brighter colors indicating higher levels of sun-induced photosynthetic florescence (NASA 2014). The higher the gross primary production, the more carbon dioxide drawn out of the atmosphere.

While new technologies for carbon removal will appear in the future, their cost effectiveness is questionable in the near term. If the US, and the rest of the world, is serious about reducing the amount of GHGs in the atmosphere, all nations will need to with existing agriculture and forestry to do so. Iowa, with its position as a leading agricultural state, its research capabilities, and its capacity for carbon storage, can be a leader in agricultural carbon management.

The amount of private sector investment in this arena already shows that the Midwestern United States is poised for carbon markets. What both producers and investors need are credible, verifiable, standardized measurements that foster additionality and efficiency, coupled with well-researched production techniques and functioning carbon markets.[1] While presently there may seem to be limited scientific consensus on which specific agricultural practices and technologies meet these goals, researchers can build on a solid knowledge base. As this report demonstrates, much is already known. While regional and federal governments around the world have been slow or haphazard in their policy actions, resulting in idiosyncratic, “Wild West” markets, the private sector is not waiting. Producers and investors need clarity and direction. If buyers and sellers in a carbon credit market are uncertain of the verity of a claim, the market will more likely fail than flourish. There is no need for government to create

[1] Additionality is the principle that carbon markets should incentivize activities that would not occur under "business-as-usual" conditions. Carbon markets are structured to motivate new activities not already mandated by regulation or resulting from common business practices. Efficiency in this context refers to providing the greatest possible carbon (CO_2) removal and/or GHG emission reduction for the lowest possible cost.

and impose a command-and-control marketplace for carbon credits, but it can set the standards and provide the regulatory guardrails so that all participants know the rules and choose the path that is best for them. Farmers want to know that the practice they have chosen is storing carbon and being measured appropriately; so do their investors. What is most needed is research and policy that lead to verifiable, credible claims.

In terms of market size, compliance markets are already substantial, and in some cases lucrative. For example, the compliance market associated with California's Cap-and-Trade Program (CARB 2015) and Low Carbon Fuel Standard (CARB 2021) are driving substantial private sector investments in renewable energy, as covered in **Chapter 2**. The market for voluntary carbon offsets is not yet large, but is growing rapidly, with a doubling expected between 2020 and 2021 (Forest Trends Ecosystem Marketplace 2021). In 2021, the market is expected to have reached over $1 billion for the first time ever. So far, however, agricultural carbon credits only comprise about 1% of this market, suggesting opportunity for expansion.

As described throughout the report, many common agricultural management practices can contribute to reductions in GHG emissions and carbon removal: cover cropping, residue management and tillage, fertilizer management, manure storage and handling, and changes in land use, to name a few. Among Iowa farmers, the most commonly cited *general* reason that a particular conservation practice or technology is not adopted is simply "lack of knowledge" about the practice. The most commonly cited *economic* reason was "pressure to make a profit makes it difficult to invest" in the practice. Education about the practices, verifiability of the additionality to carbon market participants, and better understanding of Iowa's carbon credit potential will go a long way in overcoming both of these barriers to adoption.

1.4 Summary of the Remaining Chapters in This Report

Covering carbon markets in the US, **Chapter 2** discusses carbon pricing and contract policies, regulations and industry responses that emerged over time and that have evolved into the current markets. Some states and regions as well as private sector entities are engaging in carbon markets. These markets for carbon credits, especially in agriculture, are best described as fast moving and fluid. Today's voluntary agricultural carbon programs include several entities that have developed carbon credit platforms and engaged farmers to pursue carbon projects. Juxtaposition of the various platforms and contracts reveals a large amount of variety in terms of contracts, payments, adaptation costs, verification, registration, and third-party involvement. Farmers on average state that they would like to participate in such markets but have concerns about validity, payments, and additionality. Regarding additionality, purchasers of carbon credits must be certain that the investment in new practices actually results in the removal of carbon dioxide from the atmosphere (hereafter, carbon removal) or reductions in GHG emissions. These factors are also considered in this chapter.

While policies, markets, and entities all have their idiosyncrasies, there is a fair amount of research, especially research in similar marketplaces, to inform policymakers on best practices. Much is already known about the difficulties inherent in markets that suffer from a lack of verity and trust. Before any carbon market can exist, there must be credibility allowing participants to take informed risks. Without credibility, there will be no market. Literature informs us that a role for government here is to help industries establish credibility by setting standards to enforcing rules. One advantage for Iowa is the fact that agricultural producers in the nation's second largest agricultural state by value are already engaging with the private sector in GHG reduction and carbon removal projects and market building. If Iowa is successful at market-based carbon management through its agricultural, bioenergy, and forestry practices, it can help set national standards for these practices. Iowa is poised to be an important player in agricultural carbon markets if the industries can work with local, state, and federal governments in establishing not simply an even playing field, but one where the rules are clear, the definitions and standards agreed upon, and the claims verified.

Measuring, reporting, and verification (MRV) of carbon projects and programs is currently a dynamic and multifaceted environment with many players and many competing claims. Credible MRV is of critical importance for providing the foundation for any future market. The linking of purchased carbon credits to specific practices and locations is the minimum requirement for any claim to be credible. Without credibility, the markets will likely fail. With credibility established, the potential for Iowa could be quite large. **Chapter 3** focuses primarily on the measurement component of MRV. Currently, because so many of the programs are private and voluntary, measurement approaches have varied. With the potential for agreed-upon, possibly worldwide, protocols (e.g., the 2021 COP26 United Nations Conference on Climate Change, Article 6), coupled with the eventual establishment of certification guidelines in the United States (e.g., Growing Climate Solutions Act) and requirements for accounting measures to be transparent to shareholders (e.g., US Securities and Exchange Commission forthcoming rule regarding environmental, social, and governance (ESG) pledges), measurement is becoming more standardized and transparent. Transparency is needed for market participants who want to know the rules before investing resources. Current private sector programs, their third-party verifiers, and their methods of measuring are discussed in **Chapter 3**, as are potential methods of measurement that have also been proposed in the scientific literature. Technological advancement in measurement, especially in on-farm collection of data and precision agriculture, are certain to evolve as carbon markets gain credibility and investment interest. Iowa has the potential to be at the forefront of measurement innovation because of its size, agricultural market value, potential for carbon removal, and production research and equipment engineering capabilities.

Effective carbon removal means removing carbon dioxide from the atmosphere and permanently (or at least persistently) storing it. While agricultural soils are an area of interest because of their potential to be carbon sinks, biogeochemical reactions in soils have the potential to release carbon stored there. More research is needed to determine the best ways of keeping carbon that enters the soil in the soil. Adding new practices to remove carbon or reduce GHGs will only work and be credible if those new practices lead to a verifiable change, but that change must be measured against a baseline. In **Chapter 4** of this report, researchers discuss the inventory data pertaining to carbon in the soils of the United States in general and Iowa in particular, and areas where the baseline knowledge needs improvement. Iowa State University, departments and agencies of the State of Iowa, along with collaborating agencies at the USDA have already compiled and are still compiling a vast inventory of Iowa soil organic carbon (SOC) data, with estimates of Iowa's soil inventory stretching back to the 1850s. Although more research is needed, it is well understood that pre-settlement stocks of carbon were higher than they are today. Hence, today's baseline not only tells the present amount of Iowa's SOC, but the opportunity to store additional carbon in the soil.

Chapter 4 further demonstrates that much is already known about the carbon and GHG dynamics associated with cropping systems and land management. As a result, there is a robust science base to build on as farmers learn how to manage agricultural landscapes for CO_2e while producing traditional agricultural commodities. Drainage management, tillage management, residue management, fertilizer management, cover cropping, crop rotations, perennial crops, and buffer and filter strips are well-known techniques that have been developed to address various conservation objectives such as soil health and water quality for decades. What is less well-known is how to use these techniques to effectively and efficiently manage resources to meet either net-zero or net-negative CO_2e goals. Crop production and soil management can both increase and decrease overall GHG emissions. In particular, improved management of nitrous oxide emissions from cropland will have greater impacts on CO_2e emissions than land management focused solely on storing carbon in soils, but few data exist for Iowa on the impacts of multiple management practices employed in combination. Farmers need better data on how to optimize CO_2e management alongside yield management. The potential for a large agricultural state such as Iowa to remove and store the nation's carbon depends on developing this understanding and carbon credit prices.

The livestock industry provides crucial pathways toward achieving net-zero or net-negative CO_2e goals, as discussed in **Chapter 5**. In the case of livestock, the CO_2e gases of interest are nitrous oxide and methane as opposed to carbon dioxide. Methane is especially important for two reasons. First, although comparatively short-lived, in terms of its CO_2e, methane is a potent greenhouse gas. Second, GHG reductions can be achieved through gains in feed efficiency, feed additives that reduce enteric methane production, and manure management. An important understanding is that energy lost by an animal through GHG emissions is also lost yield and, hence, profitability to the producer. Further, if manure-associated methane can be properly captured and cleaned, it is valuable as a fuel, renewable natural gas, for which there is a well-established and lucrative market. Digesters need to be easier to operate and profitable at smaller scales for the technology to become widely adopted. The impact of grazing cattle on pasture on emission is complex. Although much is known, there are numerous areas of additional research that are needed to better understand GHG emissions in grazing systems. While much is already known, producers need more information on how to measure and manage emissions, especially in grazing systems. Today, measurement is costly for such extensive management systems, and there is high variability in carbon outcomes depending on farm and techniques used. Iowa would clearly have a comparative advantage, being the nation's second largest producer of livestock-associated products by value, with large numbers of swine, beef cattle, dairy cattle, and poultry farms. Opportunities to implement and test a variety of GHG emissions reduction techniques could be explored.

Chapter 6 presents the engineering approaches to reducing GHG emissions via renewable fuels that can be used for transportation and for on-demand renewable power production. It examines transportation energy research, addressing light- and heavy-duty vehicles, aviation, and maritime transport. The chapter also provides an overview of the energy potential with a focus on Iowa's needs and capacity, and engineering technologies that could be more broadly applied to agricultural systems to reduce GHG emissions and/or remove carbon. Iowa is already a well-established leader in renewable energy generation, supporting systems with low, zero, or net negative carbon emissions such as wind, ethanol, and biodiesel. Renewable biodiesel, renewable natural gas, green hydrogen, and green ammonia are poised for demonstration and/or expansion. While some emerging technologies for carbon removal are better adapted to other regions of the country, researchers expect, for Iowa, the highest potential exists for the capture and sequestration of biogenic carbon dioxide and pyrolysis of biomass to biochar.[2] As research and technologies improve, additional options can become more feasible.

As also discussed in **Chapter 6**, a complete view of the landscape for renewable fuels development includes connections with the electric grid. Grid participants are adapting to increase renewable resource integration and energy storage, and participate under new market rules requiring equal access for these resources. The grid landscape includes robust transmission investment as the foundation of grid reliability, with multiple scales of distributed renewable power generation resources and increasingly variable loads as electrification proceeds. In simultaneously decarbonizing the grid and decarbonizing agriculture, economic synergies emerge between the two systems: grid resources may be powered by multiple fuels (wind, solar, renewable fuels), and low-cost electricity can be stored in batteries or used for distributed production of two fuels that couple power systems to renewable fuel systems: hydrogen and ammonia. Successful synergies will lead to greater economic development in Iowa and greater grid resiliency and likely lower costs, but the key will be the development of flexible, scalable, interoperable systems. There is a need for pilot projects working with agricultural producers, utilities, communities, and Iowa's universities to demonstrate the economic and technical viability and benefits and costs of such systems.

Iowa farmers are already adopting agricultural practices that contribute to CO_2e removal and storage, and many others say they intend to use them or are at least open to them. **Chapter 7** discusses what those current practices are and what practices farmers are considering. In fact, while many pose adoption or non-adoption as distinct processes, farmers actually use a continuum of practices, revealing adoption as a fluid rather than static concept. Using data gathered from USDA censuses and farmer surveys, the researchers show how the adoption of cover crops, residue and tillage management, fertilizer use, manure storage, and other management practices has changed both nationally and here in Iowa. Importantly, a survey of intended use shows which practices Iowa farmers say they are most likely to use. The research also examines reports on what barriers farmers face with "lack of knowledge" about the practice—the most commonly cited barrier.

An important aspect of farm-scale carbon management is management planning. In **Chapter 8**, researchers examine the literature and explain the management tools available to Iowa researchers and practitioners. There are a variety of "calculator" tools available and **Chapter 8** examines several of them. The COMET-Farm tool is widely used by carbon programs to calculate the baseline soil conditions and emissions and, subsequently, the net sequestration and emissions for potential adopted practices. The Agricultural Conservation Planning Framework (ACPF), and the ACPF Financial and Nutrient Reduction Tool (FiNRT), incorporate inputs from geospatial data, processing, and other analyses and through the modeling of different management techniques to produce estimates for use in conservation planning best management practices (BMPs) to meet an enterprise's conservation and financial goals. These tools are already being used by Iowa and other Midwestern producers, landowners, agencies, and

[2] Biogenic carbon dioxide that is a by-product of production streams at agricultural processing facilities can be captured and geologically sequestered. Corn ethanol plants provide a prominent example of facilities with biogenic carbon dioxide that could be captured and geologically sequestered. Pyrolysis is a moderately high-temperature industrial process that decomposes biomass into gaseous, liquid, and solid products in the absence of oxygen. The gaseous and liquid products can be upgraded into energy products. The solid product, a form of charcoal commonly known as biochar, provides numerous benefits as a soil amendment. Its recalcitrance to biological decomposition makes it an ideal carbon sequestration agent.

stakeholders. In this chapter, the results of a COMET-Farm output are presented for an Iowa State University demonstration farm near Newall, Iowa. The researchers demonstrate using management practices at the farm and a baseline 2000–2018 corn-soybean rotation with intensive tillage and the subsequent transition to a no-till system. The researchers also demonstrate the applicability of ACPF and ACPF FiNRT to carbon management in Iowa in a case study for the Upper Big Creek watershed using three conservation planning scenarios. All three tools demonstrate a variety of outcomes, not only regarding carbon and other gases, but also with respect to yield changes and costs. While management practices and carbon prices no doubt will vary, the case study demonstrates an important aspect of carbon sequestration practices in Iowa. Namely, accounting for carbon management practices is viable and that researchers, producers, and other stakeholders already have practical tools at their disposal. The conservation planning tools also show the potential for carbon management having complementary benefits to water quality management.

Process-based agroecosystem models provide mathematical representations of one or more biophysical processes that govern the stocks and flows of energy and materials within ecosystems. They are primarily constructed as scientific endeavors, but are also used to inform decision-making and as a platform from which to build calculators for credit estimation. Researchers in **Chapter 9** summarize eight different process-based agroecosystem models that have undergone substantial development and have been applied in the Corn Belt, including Agro-IBIS, APSIM, CENTURY, Cycles, DayCent, DLEM, DNDC, and EPIC. In this chapter, the Dynamic Land Ecosystem Model (DLEM) is used to examine changes in Soil Organic Carbon (SOC) stocks and how carbon dioxide and nitrous oxide have ebbed and flowed across Iowa from the 1850 to 2019. The model predicts a SOC deficit of 700 million Mt carbon for Iowa today compared to the pre-EuroAmerican settlement period. The model also predicts that GHG fluxes across Iowa prior to EuroAmerican settlement were dominated by carbon dioxide. Specifically, in the 1900s the majority of Iowa was a minor net carbon dioxide source, although larger net carbon dioxide emissions were located in a few smaller regions in central and northern Iowa. Interestingly, by the 2010s, large areas within Iowa turned into small carbon dioxide sinks, although the state remained a net carbon dioxide source. In contrast, nitrous oxide emissions in the 2010s were significantly elevated throughout Iowa relative to historical periods. The results of the DLEM model simulation indicate that the increases in net GHG fluxes across Iowa between 1950 and 2019 were mainly driven by enhanced nitrous oxide emissions. Both soil carbon and GHG modeling results imply a robust potential for Iowa to garner carbon credits through changes in agricultural management.

Most carbon supply chain models follow aspects of the International Organization for Standardization (ISO) in their life cycle assessment (LCA). **Chapter 10** provides the background for these standards, which predate carbon assessment by decades. Industries and government agencies agreed long ago that there was a need for consistency across countries and industries in a variety of manufacturing and energy delivery systems for many reasons, including accurate measurement of environmental impacts. Carbon sequestration projects are, in that sense, just the latest process to be systematized. After examining the rationale for these systems, **Chapter 10** discusses the life cycle impacts of technologies that could put Iowa at the forefront for carbon removal. An important aspect of this analysis is that the benefits (in terms of reduced carbon) must also be measured against the costs. The analysis in this chapter shows that, for several technologies, Iowa has a substantial potential for cost-effectively contributing to carbon removal. There are opportunities for academic institutions, national laboratories, and private organizations to improve lifecycle inventory data that captures Iowa's unique resources and agricultural and energy industries. An Iowa-based database of the Greenhouse Gases, Regulated Emissions, and Energy Use in Technologies (GREET) model could be developed in partnership with compliance consulting firms and state institutions, and would go a long way toward supporting GHG emissions reduction and carbon removal efforts.

While each of the report's chapters summarize specific research needs associated with the areas covered, the overarching need for systems research and demonstration projects emerges when looking at the whole. Comprehensive life cycle assessment of emerging low-carbon and carbon negative engineering technologies based on data collected from field studies of Iowa agroecosystems are currently lacking. Systems analyses could provide understanding about potential new synergies between agricultural and energy sectors enabled by carbon markets and renewable electric power. Demonstration projects located in realistic environments are needed to evaluate and improve the viability of novel agricultural practices and emerging engineering technologies used in combination. Innovative partnerships among farmers, industries, communities, and Iowa's universities could move these efforts forward at a faster pace than if sectors were operating independently. A second emergent theme is, regardless of how it's monetized, carbon will continue to provide substantial value to farms, communities, and the state for long to come.

1.5 Units and Unit Conversion Information

We generally use English units of measurement in this report because of the broad, largely Iowa-based, audience we are writing for. We make an exception throughout by reporting GHGs according to their CO_2e and SOC in metric tons to be consistent with carbon markets. While this results in the somewhat awkward mixing of English and metric measurement systems (e.g., metric tons per acre), we expect it is most communicative to our audience. We provide Table 1.1 so readers who are more familiar with other units are easily able to convert values. Values are generally reported to two significant digits, unless there is a reason to report additional digits (e.g., with dollars and cents).

Table 1.1. Common unit conversions.

Metric Unit	Abbreviation	Alternate Metric Values		English Units
1 Metric ton	(Mt)	1000 kg	10^6 g	2204 lb
1 Million Metric tons	(MMt)	10^9 kg	1 Pg	1.1M tons
1 Petagram	(Pg)	1 MMt	10^{15} g	1.1M tons
1 Centimeter	(cm)	0.01 m	10 mm	0.393 in
1 Meter	(m)	100 cm	1000 mm	3.28 ft
1 Kilometer	(km)	1000 m	100,000 cm	0.621 mi

We focus our assessment on three GHGs that are affected by agricultural land management and supply chains: carbon dioxide, methane, and nitrous oxide. We use multiple global warming potential (GWP) values in this report, depending on the publication being cited or what the respective authors used in their work, also recognizing that different GWPs for the same gas are typically within the range of uncertainty of their values. Table 1.2 provides information needed to translate values to other GWP measurement systems. **Appendix 11.2** provides for further information on computing GWP.

Table 1.2. Basic chemistry and climate science information about greenhouse gases considered in this report. GWP = Global Warming Potential; C = carbon; N = nitrogen.

Greenhouse Gas	Symbol	Molecular Weights (g/mole)	AR4 GWP-100	AR5 GWP-100	%C or %N
Carbon dioxide	CO_2	44	1	1	27
Methane	CH_4	16	25	28-34	75
Nitrous oxide	N_2O	44	298	265-298	64

Other conversions that may be of interest include:

- Soil Organic Matter (OM) to Soil Organic Carbon (SOC): 0.5 x OM% = SOC% (approximate for most soils), and
- Carbon dioxide (CO_2) to Soil Organic Carbon (SOC): 1 g CO_2 x 12 g C / 44 g CO_2 = 0.27 g SOC.

1.6 Abbreviations

We make limited use abbreviations in this report to improve accessibility and readability of information contained within. In some cases, however, using abbreviations actually improves accessibility and readability. This is the case for several terms used ubiquitously throughout the report, and their abbreviations are well known. It is also the case for terms and the names of models that are very long when written out. In these cases, we define all abbreviations at their first use in each chapter. A complete list of abbreviations used in the report follows.

ACPF	Agricultural Conservation Planning Framework
APSIM	Agricultural Production Systems Simulator
BECCS	Bioenergy with Carbon Capture and Sequestration
BMP	Best Management Practice
CCS	Carbon Dioxide Capture and Sequestration
CDL	Cropland Data Layer
CO_2e	Carbon Dioxide Equivalent
CaRPE	Carbon Reduction Potential Evaluation Tool
DayCent	Daily CENTURY model
DLEM	Dynamic Land Ecosystem Model
DMI	Dry Matter Intake
DNDC	Denitrification-Decomposition model

EPIC	Environmental Policy Integrated Climate
ESG	Environmental, Social, and Governance
FiNRT	Financial and Nutrient Reduction Tool
GHG	Greenhouse Gas
GREET	Greenhouse Gases, Regulated Emissions, and Energy Use in Technologies
GWP	Global Warming Potential
HEFA	Hydro-processed Esters and Fatty Acids
HOLCF	High-Octane, Low-Carbon Fuel
IDALS	Iowa Department of Agriculture and Land Stewardship
IDNR	Iowa Department of Natural Resources
IFRLP	Iowa Farm and Rural Life Poll
INRS	Iowa Nutrient Reduction Strategy Survey
LCA	Life Cycle Assessment
LOI	Loss On Ignition
MCF	Methane Conversion Factor
MRV	Measurement, Recording, and Verification
NASS	National Agricultural Statistics Service
OpTIS	Operational Tillage Information System
RNG	Renewable Natural Gas
SAF	Sustainable Aviation Fuels
SOC	Soil Organic Carbon
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
WASCOB	Water and Sediment Control Basin

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Chapter 2. Carbon Markets, Past and Present

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2.1 Highlights

- Voluntary agricultural carbon markets in the United States can be viable, and many private firms are establishing a presence.
- The lack of standards and guidance confuses farmers and carbon credit buyers, hindering market development.
- Iowa farmers are interested in carbon markets, but uncertain about them.
- Challenges to a robust marketplace for agricultural carbon credits are well understood and many hinge on the credibility and cost of the certification process.

2.2 Background

Improved agricultural practices can help mitigate climate change by reducing emissions created by agriculture and other sources. Farming practices that are commonly accepted as methods to reduce or remove atmospheric carbon include tillage management (reduced till, no-till, strip-till), improved cropping practices (cover cropping, extended crop rotations, diversification of cropping system, including perennial crops), improved nitrogen efficiency (nitrogen inhibitors, split applications, and in-season applications), and grazing management. There is a large potential for developing the supply side of carbon credits in agriculture because there is ample room to improve the adoption of practices that reduce or remove atmospheric carbon. Only 3.9% of total cropland area in the continental United States was planted to cover crops, and only 26.4% was in no-till systems in 2017 (US Census of Agriculture 2019), while nitrogen application rates were above recommended rates in 36% of corn acres, 19% of cotton acres, 22% of spring wheat acres, and 25% of winter wheat acres in 2010/11 (Wade et al. 2015). Sellars et al. (2020) reported that 67% of the fields surveyed in Illinois received a nitrogen application above the Maximum Return to Nitrogen (MRTN) recommended rate.

Implementing new agricultural practices (additionality) that sequester carbon for extended periods of time (permanence) is challenging for farmers. Practice changes are typically associated with increased short-term costs (caused by additional inputs, labor, learning curve, yield losses) and uncertain long-term benefits (increased long-term yields and resilience, soil health, land value). The uncertainty of long-term benefits refers to both the magnitude and the timing of the private benefits accrued by the farmer. For example, no-till could reduce crop productivity, particularly in cooler and/or wetter climatic conditions as a result of the surface residues and lower soil temperatures (Ogle et al. 2012) and require more than five years to reduce soil erosion and sediment loss to water and wind and increase the soil water-storage capacity (Toliver et al. 2012). Furthermore, the economic incentives offered by carbon programs to induce farmers to implement improved practices can vary substantially across programs in terms of contract length, amount and timing of payment, and multiple other factors (Plastina and Wongpiyabovorn 2021).

In a market economy, prices carry signals for both producers and consumers. When greenhouse gas (GHG) emissions are costless to the emitter, there is little incentive to incorporate the environmental and social costs imposed by the emissions (i.e., the externalities) into the emitter's decision-making process (i.e., internalize the externalities). The World Bank's Carbon Pricing Dashboard (World Bank Group 2021) lists 65 carbon pricing programs around the world, covering 21.5% of GHG emissions worldwide and generating $53 billion in revenue. The menu of carbon pricing programs includes mandatory and voluntary emissions trading systems, GHG taxes, and combinations of emissions trading systems and taxes. Because GHGs have different global warming potentials (GWPs), all pricing programs express the covered GHG into carbon dioxide equivalent (CO_2e) units.

2.3 Carbon Pricing in the United States

In the United States, demand for carbon offsets generated in the agricultural sector was until recently driven by the derived demand for offsets from three emission-trading systems: the Chicago Climate Exchange, the California Cap-and-Trade Program, and the Regional Greenhouse Gas Initiative.[3] In recent years, there has been an explosion of voluntary programs to generate saleable CO_2e emission reductions and net CO_2e removals, driven mainly by corporate environmental, social, and governance (ESG) pledges from major corporations. The following sub-sections summarize the workings of carbon pricing programs in the United States (adapted from Wongpiyabovorn et al. 2021).

2.3.1 The Chicago Climate Exchange (CCX)

In 2003, the CCX was established as the world's first and North America's only active voluntary GHG emissions cap-and-trade program. It was voluntary in the sense that participants voluntarily chose whether to participate in the CCX, but participants were legally obliged to achieve their annual emission reduction target. The program targeted six GHG emissions: carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), and sulfur hexafluoride (SF_6). It included participants from the United States, eight Canadian provinces, and sixteen other countries, while incorporating offset projects worldwide. Participants established their own GHG emission baselines and were allocated annual emission allowances that ranged from 99% of their emission baseline in 2003 to 94% of their baseline in 2010. Members who reduced emissions below their targets had surplus allowances, known as exchange allowances, to sell or bank. Members who emitted GHG above their targets complied by purchasing CCX Carbon Financial Instrument Contracts (CFICs), each representing 100 metric tons of CO_2e (Mt CO_2e).

CFICs included exchange allowances and exchange offsets. Exchange offsets were generated by qualifying offset projects on the basis of removal, destruction, or reduction of GHG emissions. All CCX

[3] The 2009 American Clean Energy and Security Act proposal intended to establish an emission cap-and-trade program that would have covered seven major greenhouse gases from large emitters, petroleum fuels producers and importers, and gas distributors (PEW Center on Global Change 2009). Using offsets from agriculture and forestry sources would have been allowed to meet compliance requirements. However, the bill did not materialize, partly because of lack of support in the Senate and the effects of high unemployment stemming from the Great Recession (Weiss 2010).

offsets were issued on a retrospective basis with the CFIC vintage (i.e., the year when the offset was issued) applying to the program year when the GHG reduction took place. Projects underwent third-party verifications, and verification reports were inspected for completeness by the Financial Industry Regulatory Authority (FINRA). The only agricultural offset projects that qualified for the CCX were based on methane collection and soil carbon sequestration. The minimum scale to trade carbon offsets in the CCX market was 10,000 Mt CO_2e (equivalent to 100 CFICs), which roughly translates into 25,000 ac in conservation practices (Ribera and McCarl 2009). Consequently, most agricultural projects were managed by aggregators that charged 8-10% of the value of carbon offsets at market price on a yearly basis (Ribera et al. 2009). Forestation, forest enrichment and conservation, and urban tree planting also qualified for generating CCX offsets. The scale required to supply carbon offsets to the CCX severely limited interest from the agricultural sector and small forest landowners.

Although the price of carbon offsets traded in the CCX peaked at $7.40 per Mt CO_2e in May 2008, it plummeted to 10 cents per Mt CO_2e in August 2010 (Griesinger 2010). Comfortable baselines, unambitious emission reduction targets, lack of a minimum price on CFICs, and investments in new and cleaner technologies by CCX members contributed to the ceasing of the trading platform in 2010. The problem of the CCX was that the verified emission reductions exceeded the compliance requirement, resulting in an oversupply of carbon offsets (ICE 2011).

2.3.2 The Regional Greenhouse Gas Initiative (RGGI)

In 2005, the RGGI was established as a cooperative effort among the states of Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Rhode Island, Vermont, and Virginia to cap and reduce carbon dioxide emissions from fossil fuel power plants with an output exceeding 25 megawatts. The compliance obligation started in 2009 with a cap of 170 million Mt CO_2, which declined to 82.6 million Mt CO_2 by 2014, and was further reduced by 2.5% per year to reach 72.8 million Mt CO_2 in 2019. However, the cap was lowered in 2014 to account for the surplus of allowances previously accumulated. The adjusted cap amounted to 52 million Mt CO_2 in 2019. New Jersey left the program in 2012 and rejoined it in 2020 and Virginia did not fully participate until 2021. The adjusted cap increased to 67 million and 91 million Mt CO_2 in 2020 and 2021, respectively. The regional cap will gradually decrease to total a 30% emission reduction by 2030 relative to the 2020 emission level.

Allowances are distributed quarterly via regional auctions. The first auction of RGGI allowances took place in September 2008. To prevent extreme allowance price fluctuations, a Cost Containment Reserve (CCR) and an Emissions Containment Reserve (ECR) were implemented in 2014 and 2021, respectively. The CCR is the mechanism to hold allowances in reserve and sell them if allowance prices exceed the trigger price. The CCR trigger price started at $4 in 2014 and climbed to $13 in 2021. The trigger price will increase by 7% annually thereafter. The size of the CCR is 10% of the regional cap each year. The CCR allowances were sold twice in March 2014 and September 2015 at prices of $4 and $6.02, respectively. The ECR allows states to withhold up to 10% of their annual budget if prices fall below the trigger price ($6 in 2021). The ECR trigger price will also increase by 7% annually. Although the trigger price of the CCR and ECR are increasing at the same rate, the 2021 price of the CCR is higher. As a result, the gap between these two trigger prices will widen. In sum, the CCR acts as a ceiling to allowance prices, and the ECR acts as a floor.

While agriculture and forest emissions are not directly regulated in the RGGI, carbon offsets from methane capture and destruction, and from carbon removal through afforestation can be purchased by power plants to be used against their excess carbon dioxide emissions. The use of offsets is limited to 3.3% of a power plant's total compliance obligation. The offsets are also issued on a retrospective basis and require third-party verification. The RGGI carbon dioxide Allowance Tracking System (RGGI 2021) only lists one project as an authorized source of offsets that has produced 48,540 Mt CO_2e through landfill methane capture and destruction in Maryland since 2017. In comparison, 156,464,910 Mt CO_2 were auctioned off over the control period between January 1, 2018 and December 31, 2020, at an average price of $6.05 per Mt CO_2.

According to the Acadia Center (2019), the carbon dioxide emissions from power plants in RGGI states declined by 47%, from 121 million Mt CO_2 in 2008 to 64 million Mt CO_2 in 2018. Emissions have typically been lower than the cap throughout the program. Accordingly, the RGGI allowances were sold at the reserve price in 11 auctions between 2010 and 2012, and the allowance prices were below $4.41 per Mt CO_2 until 2014. The average auction clearing prices increased to $4.87 in 2018, $5.98 in 2019, and $7.07 per Mt CO_2 in 2020. The allowance price jumped to $10.25 per Mt CO_2 in the November 2021 auction. Throughout the life of the initiative, carbon dioxide offsets were not widely used, representing less than 0.1 million allowances, compared to more than 1 billion excess allowances sold at auctions.

2.3.3 The California Cap-and-Trade Program (CCTP)

The CCPT, launched in 2013, places a cap on GHG emissions from the state's power, industrial, and transportation sectors. Facilities that emit more than 25,000 Mt CO_2e per year are required to comply with the cap-and-trade program. The program covers three GHGs—carbon dioxide, methane, and nitrous oxide—accounting for about 80% of the state's GHG emissions.[4] The California Air Resource Board (CARB) established a cap at 2% below the forecasted 2012 emissions level in 2013, declining at an annual rate of 2% in 2014 and 3% from 2015 to 2020 (CARB 2015). The allowance budget will decrease by 13.4 million Mt CO_2e from 2021 to 2030, and by 6.7 million Mt CO_2e per year starting in 2031 (CARB 2019).

California's allowances are distributed via free allocation and auction. Facilities receive free allocation at about 90% of average emissions, updated yearly based on production data. In addition, participants are allowed to bank their unused allowances, subject to holding limits, for future compliance. However, borrowing from future allowances is not permitted. California's program linked with Québec's cap-and-trade program in 2014 and Ontario's program in 2018, although the latter linkage was short-lived. The linkage allows the use of allowances issued in Québec's program to meet compliance obligations in California and vice-versa.

Allowances are auctioned-off under two programs: in the Current Auction, allowances for the current year are traded; and in the Advance Auction, allowances for the third year into the future are traded, up to a volume equal to 10% of the combined allowance budgets for that year. In 2016 and 2017, auction settlement prices ranged between $10 and $12.73 per Mt CO_2e. In 2018, allowance prices averaged $14.91 per Mt CO_2e in the Current Auction and $14.82 per Mt CO_2e in the Advance Auction. Average prices for both Current and Advance Auctions remained above $16 in 2019 and above $17 in 2020. In 2021, the auction reserve price, which is the minimum price at auction, was set at $17.71 per Mt CO_2e, and it will increase annually by 5% plus inflation (International Carbon Action Partnership 2021a). In the

[4] California uses global warming potential conversion factors for methane and nitrous oxide into CO_2e of 25:1 and 298:1, respectively.

August 2021 auction, the 2021 vintage allowance price was $23.30 per Mt CO_2, and the 2024 vintage allowance price was $23.69 per Mt CO_2.

CARB holds a number of allowances in the Allowance Price Containment Reserve ("Reserve") to sell following a quarterly auction when a settlement price is higher than or equal to 60% of the lowest Reserve tier price. In 2021, allowances in the Reserve will be offered at two tier prices: $41.40 and $53.20, and the prices will increase by 5% plus inflation each year. The reserved allowances are also sold in the third quarter of the year before the annual compliance deadline on November 1st.

A regulated facility can use offsets from unregulated sectors within the United States to meet up to 8% of a facility's compliance obligation until 2020, up to 4% from 2021 to 2025, and up to 6% from 2026 to 2030. CO_2e offsets are issued on a retrospective basis, and the generating project must be verified by an independent third-party accredited by CARB. The program only allows agricultural offsets from capturing of methane from livestock manure and rice (Murray 2015). As of October 12, 2021, total offsets issued throughout the life of the program amounted to 228 million Mt CO_2e. According to the statistics from CARB, only 3.5% of those offsets came from livestock projects, and none from rice cultivation projects, whereas forestry projects generated 82% of total offsets (via reforestation, improved forest management, and avoided conversion).

California's total GHG emissions decreased by 7.3% between 2012 and 2019. By the end of 2014, the state had reduced GHG emissions by 8.3 million Mt CO_2e, compared to 2012 levels, after accounting for 12.7 million Mt CO_2e in offsets. In the second compliance period (2015-2017), emissions declined by 18.5 million Mt CO_2e, after accounting for 62.7 million Mt CO_2e in offsets. In 2018-2019, emissions were reduced by 6.3 million Mt CO_2e, after accounting for 10.8 million Mt CO_2e from offsets. In addition, 5.8 million Mt CO_2e from offsets were used to meet compliance in Québec.

2.3.4 Voluntary Carbon Credits

Currently, carbon credits from some agricultural practices, such as cover cropping and fertilizer use reductions, cannot be used to comply with emission targets in mandatory United States carbon markets. The demand for agricultural carbon credits in voluntary markets is expected to stem mostly from the implementation of "net-zero" GHG emissions pledges by more than 1,500 companies and 120 nations (Black et al. 2021). Corporations as varied as IBM, JP Morgan Chase, Boston Consulting Group, Dogfish Head Craft Brewing, Shopify, Anheuser-Busch, and Barclays have announced entering into agreements with Indigo Ag to finance the generation of carbon credits. The largest and most detailed announcement so far was made by Microsoft, indicating a commitment to remove 1.3 million Mt CO_2e from the atmosphere (equivalent to about 11% of the annual emissions from its value chain) through afforestation projects in Peru, Nicaragua, and the United States; soil regeneration across farms in the United States; and industrial sequestration of carbon dioxide from the air and injection into the ground where it mineralizes (Joppa et al. 2021). Microsoft specified that less than half of the purchased carbon credits will be certified to officially compensate for its direct emissions (i.e., turn them into carbon offsets), reminding us that certified carbon offsets and non-certified carbon credits will compete for market share in voluntary markets and attract different prices. In addition, because the demand for certified offsets in mandatory markets is driven by regulatory obligations while the demand for voluntary credits is driven by softer targets, the prices for the latter will tend to be lower than the prices for the former.

The current supply of agricultural carbon credits in the United States is very limited, but with the recent advent of numerous voluntary carbon programs offering farmers a long menu of options to generate carbon credits, this could rapidly change under the right conditions, while generating an additional income stream for program participants. A survey of 11 private voluntary programs indicates that tillage management (reduced till, no-till, strip-till), improved cropping practices (cover cropping, extended crop rotations, and diversification of cropping system, including perennial crops), grazing management, and improved nitrogen efficiency (nitrogen inhibitors, split applications, and in-season applications) are the most commonly accepted farming practices to generate agricultural carbon credits (Plastina and Wongpiyabovorn 2021). Given that only 3.88% and 26.35% of the continental United States cropland is planted to cover crops or is in no-till systems (Sawadgo and Plastina, forthcoming), and that nitrogen application rates are above recommended rates in 36% of corn acres, 19% of cotton acres, 22% of spring wheat acres, and 25% of winter wheat acres (Wade et al. 2015), there seems to be a large potential for developing the supply of agricultural carbon credits.

It must be noted that while the USDA administers several voluntary conservation programs—including the Conservation Reserve Program (CRP), the Environmental Quality Incentives Program (EQIP), the Conservation Stewardship Program (CSP), and Conservation Technical Assistance (CTA)—none of these are tailored towards fighting climate change or removing carbon from the atmosphere. However, they indirectly incentivize carbon removal by supporting conservation activities to improve water and air quality, increase soil health, and reduce soil erosion.

2.3.5 Carbon Taxes

While emissions trading systems limit GHG emissions through maximum allowances and minimum prices, governments can also generate disincentives for the emission of GHG via taxation. Carbon taxes can be imposed on domestic production to incentivize GHG emission reductions in the country levying taxes, or on imports of products with large GHG footprints to incentivize emission reductions in the exporting countries.

In the United States, the Joint Committee on Taxation and the Congressional Budget Office projected that a broad-based carbon tax, starting at \$25 per Mt CO_2e in 2017 and rising at 2% above inflation rates, would have raised \$1 trillion over its first decade (Congressional Budget Office 2016). Projections by the Brookings Institution (Barron et al. 2019) suggest that a \$25 tax per Mt CO_2e that rises by 1% per year would reduce emissions by 17–38% by 2030, compared to 2005 levels. A \$50 per Mt CO_2e tax, increasing by 5% per year, would cause GHG emissions to decline by 26–47% relative to 2005 levels, equivalent to up to 90% of the reductions needed to achieve the United States goal under the Paris Agreement. Resources for the Future (Hafstead et al. 2021) predicts that a \$15 tax per Mt CO_2e levied on oil and gas producers starting in 2023 and increasing by 5% per year would reduce emissions to about 40% of the 2005 levels by 2030. Under current policies, the United States is projected to reduce emissions by 20–26% by 2030, compared with 2005 levels (Pitt et al. 2021), well below the United States target of almost cutting in half GHG emissions by 2030 in the Paris Agreement. However, a carbon tax on domestic production does not seem to be on the agenda of policymakers in the United States.

On July 19, 2021, Democratic lawmakers in the United States proposed a carbon border tax for imported petroleum, natural gas and coal, and other imported products that have a large carbon footprint such as aluminum, steel, iron, and cement. The border tax would apply to about 12% of United States imports and would raise between $5 billion and $16 billion per year, starting in 2024. However, the proposal faces steep resistance in Congress (Friedman 2021).

2.4 Voluntary Agricultural Carbon Programs

This section, adapted from Plastina (2021), presents a simplified description of how data, payments, and methods flow in voluntary ag carbon programs, with the intention to highlight who will have access to data on farm practices, who is the most likely buyer of carbon credits for each carbon program, who controls the methodology that will be used to translate farm practices into carbon credits, and who issues payments to program participants (farmers, project developers, carbon program, verifiers, registries, soil labs, and data platforms). The analysis is presented in flowcharts in which arrows are pointing in the direction that data, payments, methods, and carbon credits move within each carbon program. The report (Plastina 2021) is a living document that will be adjusted as new information becomes available. For a detailed analysis of eleven voluntary carbon programs, see Plastina and Wongpiyabovorn (2021), which also is a living document. Currently, all voluntary agricultural carbon programs are evaluating protocols to generate, verify, and issue carbon credits based on agricultural practices, but only Indigo Ag has a methodology approved by a Voluntary Market Registry—the Methodology for Improved Agricultural Land Management (VM0042), approved by Verra.[5]

Figure 2.1 describes a traditional carbon offset generation system, with the following nine figures showing voluntary carbon programs currently operating in the United States presented in alphabetical order. It is important to understand the workings of the existing markets for carbon offsets before exploring the newer carbon programs.

A major difference between the traditional carbon offsets and the carbon credits generated in the newer, voluntary carbon programs resides in the potential gap in their perceived qualities. A carbon offset is considered a top-quality token for 1 Mt CO_2e removed through practices that adhere to trusted protocols ensuring additionality and permanence, which are verified by an independent third party, certified, and registered with a unique serial number into a secure ledger called the “registry.” The registry is typically linked to a network of registries that serves as a clearinghouse of information on carbon credits (issued, unsold, sold, and retired) to avoid duplications and enhance transparency. When an owner of a carbon offset uses it to compensate for emissions of CO_2e somewhere else, the serial number is retired from the registry (and the transaction is transparent to the clearinghouse).

A carbon credit may or may not be perceived as being of comparable quality to a carbon offset. If carbon credits are perceived as being of lower quality than carbon offsets, they would tend to attract lower market prices than offsets. The perceived quality of carbon credits is expected to be higher when verification and issuance are external to the carbon program and lower when those critical processes are internal to the carbon program. By illustrating whether verification and issuance are external or internal processes to the carbon program, the following subsections provide some basis to anticipate

[5] Voluntary Market Registries include the American Carbon Registry (ACR); the Gold Standard Registry and the Climate Action Reserve (CAR) managed by APX Inc.; the Social Carbon Registry and the Plan Vivo Registry managed by Markit; the Verified Carbon Standard (VCS) Registry and the Climate, Community, and Biodiversity Standards (CCBS) Registry, managed by Verra.

differences in the perceived qualities and resulting prices for agriculture carbon credits issued by different programs.

2.4.1 Traditional Carbon Offset Generation

As discussed in Section 2.2., compliance markets in the United States offer limited opportunities to farmers and ranchers to register carbon offsets generated via agricultural conservation practices. When a farmer owns a serial number issued by a registry, they can sell the carbon offsets associated with that serial number to any potential buyer from the regulated industries. Only top-quality carbon offsets are traded in compliance markets, requiring additionality, permanence, project design and implementation according to registry protocols, independent third-party verification, and in some cases additional approval by a regulatory body. The CCTP is a compliance market where some top-quality carbon offsets could be sold.

It typically takes several years from project design to carbon offset issuance, and the party implementing practices to generate carbon offsets usually enlists the collaboration of project developers to navigate the process. Because of the scale of the projects and the time lag between implementation of practices and issuance of offsets by registries, most projects are financed through emission reduction purchase agreements (ERPAs), according to which an investor purchases the right to own the serial number of the registered carbon offsets and makes front-loaded payments to project developers and participants (see Figure 2.1). Given the risks involved in financing these projects, the cost to investors of carbon offsets financed through ERPAs is much lower than the price of (issued) carbon offsets in the spot market.

The investing entity uses the serial number from the registry in its GHG accounting system to compensate its emissions and "retires" the credit (making it no longer available for resale). The entity will also communicate the reduction of its GHG footprint to customers, owners, and stakeholders through its ESG reports. Farm production data is shared with project developers, independent verifiers, and registries. Payments are distributed over the life of the project.

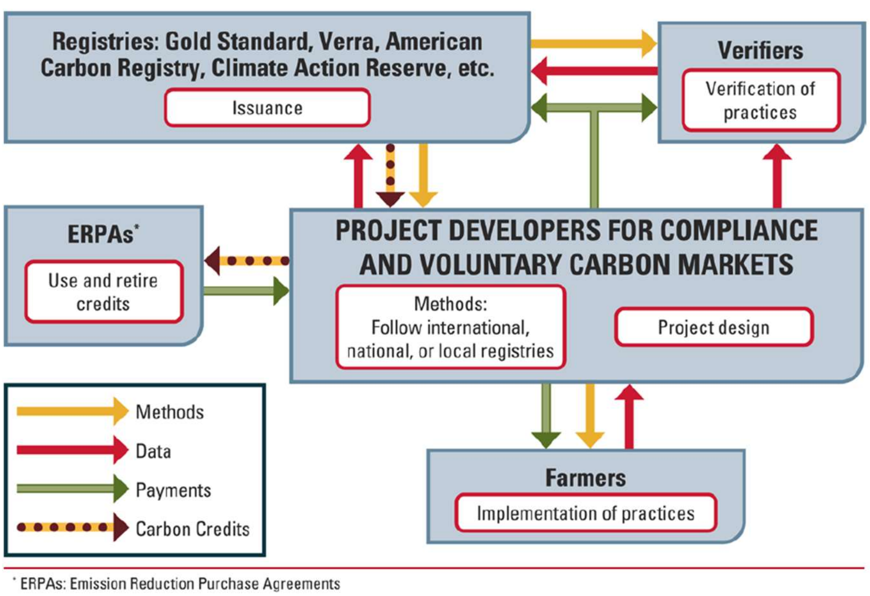


Figure 2.1. Traditional carbon offset generation.

2.4.2 Agoro Carbon Alliance

The Agoro Carbon Alliance contracts and supports farmers directly to generate carbon credits through regenerative practices, including reduced/no-till, planting cover crops, pastureland management, and nitrogen management, with further methodologies under review. Its methodology to translate agricultural practices into carbon credits is based on protocols from the Verra and Gold Standard registries (Figure 2.2). Practices implemented by farmers are registered online in the Agoro Platform and independently checked by accredited verifiers. Soil tests are mandatory and paid for by Agoro Carbon. The associated registries will issue serial numbers for carbon credits to Agoro, which in turn transfers them to buyers post-sale. Farmers have two payment options: after verifications or annual forward payments based on estimates. Farm production data is shared with project developers, Agoro, verifiers, and the respective registry. To participate in the Agoro Carbon Alliance, farmers must enroll at least 500 acres in the program for 10 years.

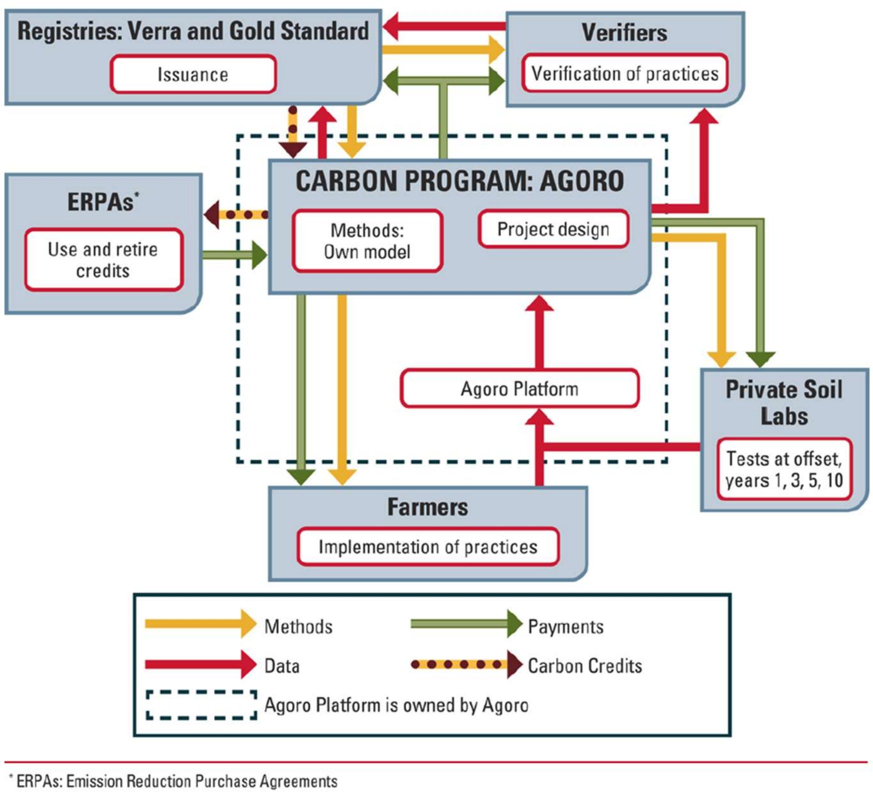
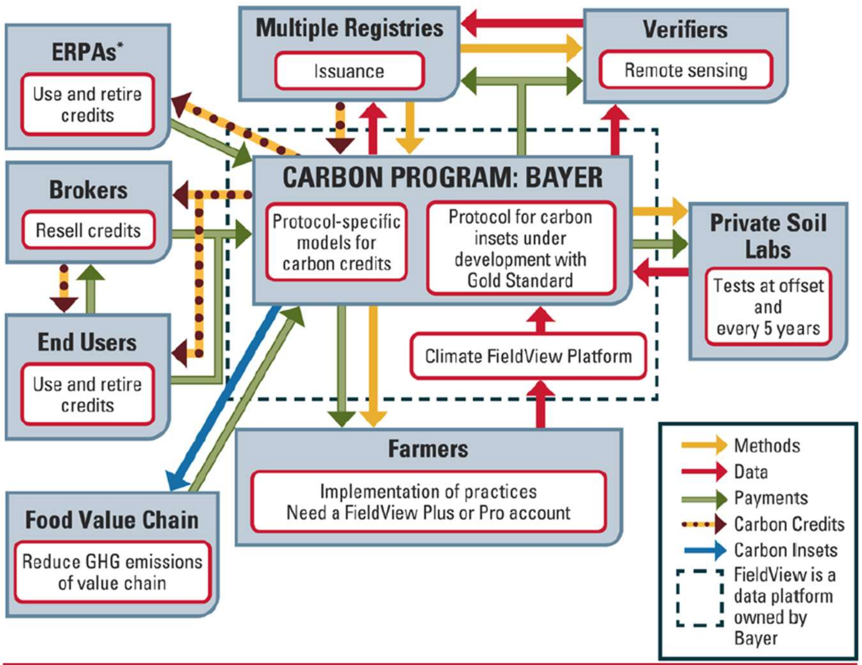


Figure 2.2. Carbon credit generation through Agoro Carbon Alliance.

2.4.3 Bayer Carbon

Bayer Carbon finds investors to finance projects through ERPAs and pays farmers $3 per acre (ac) per year to implement no-till/strip-till, $6 per ac per year to plant cover crops, and $9 per ac per year to implement both practices. Payments for implemented practices could increase (not decrease) depending on revenue obtained at credit sale. Bayer Carbon allows enrollment of practices that began as early as 2012 and offers up to five years of historic back pay after verification and validation. The methodology to quantify and issue carbon credits is under development, in collaboration with multiple registries (Figure 2.3). Farmers contract directly with Bayer Carbon and share their production data through the Climate FieldView Platform (owned by Bayer). Farmers must have a Climate FieldView PLUS subscription, which is available for free via BayerPLUS. Soil tests are mandatory at offset and every five years for the majority of the acres, and test costs are covered by Bayer Carbon. Depending on the final institutional arrangement for credit issuance and practice verification, production data may or may not be shared with actors external to Bayer Carbon only for purposes stated in the agreement, on a need-to-know basis. Payments are made on an annual basis after remote verification and validation, within one year of practice completion. Bayer Carbon offers participating farmers access to premium low-carbon grain markets. To participate in Bayer Carbon, farmers must enroll at least 10 ac in the program for five years.



* ERPAs: Emission Reduction Purchase Agreements

Figure 2.3. Carbon credit generation through Bayer Carbon.

2.4.4 CIBO

CIBO is its own registry and marketplace, and applies a methodology to translate agricultural practices into carbon credits that is based on the SALUS model (owned by Michigan State University). Project developers can be internal or external to CIBO (Figure 2.4). Practices implemented by farmers are registered online in the CIBO Plus Land Platform. Verification relies on remote sensing and is internal to CIBO. Soil tests are required only if the farm is audited, and CIBO issues the payments to soil labs. CIBO issues a serial number for carbon credits generated in a project and assigns 80% of the credits to the farmer and retains 20% of the credits as fees. Farmers sell their carbon credits through CIBO's online marketplace to end-users and brokers (who ultimately resell them to end users) and receive full monetary compensation from which fees to external project developers (if any) are paid. CIBO issues payments to soil labs. Farm production data is shared with project developers and CIBO. Payments start flowing into the system when a sale of (issued) carbon credits occurs. CIBO offers annual contracts, as well as five-year and 10-year contracts to farmers, and does not require a minimum enrollment acreage.

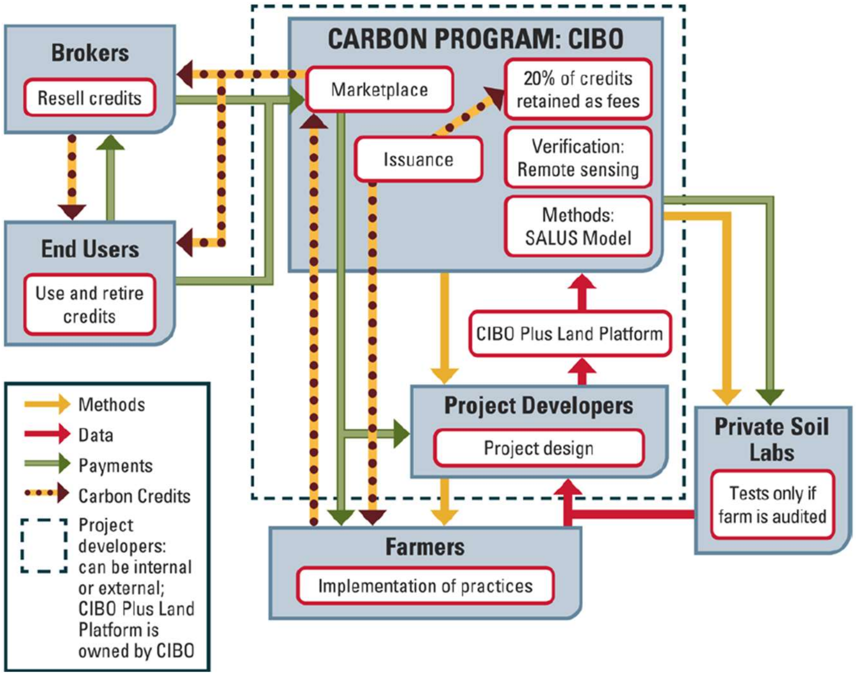
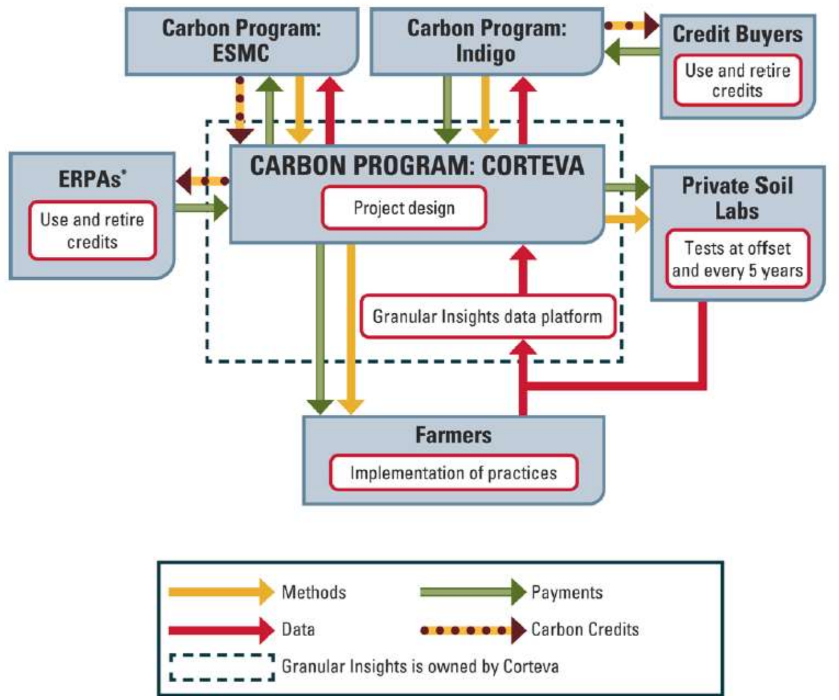


Figure 2.4. Carbon credit generation through CIBO.

2.4.5 Corteva Agriscience

Corteva Agriscience contracts directly with farmers to produce carbon credits (Figure 2.5). Corteva partners with measuring, reporting, and verification (MRV) companies, such as ESMC and Indigo Ag, to quantify and certify carbon credits through registry-approved protocols, including SustainCERT (ESMC) or Verra/CAR (Indigo). Farmers input their practices into Granular Insights, Corteva's free digital tool. These practices are submitted to carbon registries for certification and are verified through remote sensing and random site visits. Soil tests are mandatory every five years. Verifiers issue carbon credits to ESMC and Indigo, who sell credits to investors. Corteva transfers 75% of carbon credit sale to farmers, and payments are distributed over the life of the project. Corteva Agriscience offers five-year contracts, with an annual option to opt out, and does not require a minimum enrollment acreage.

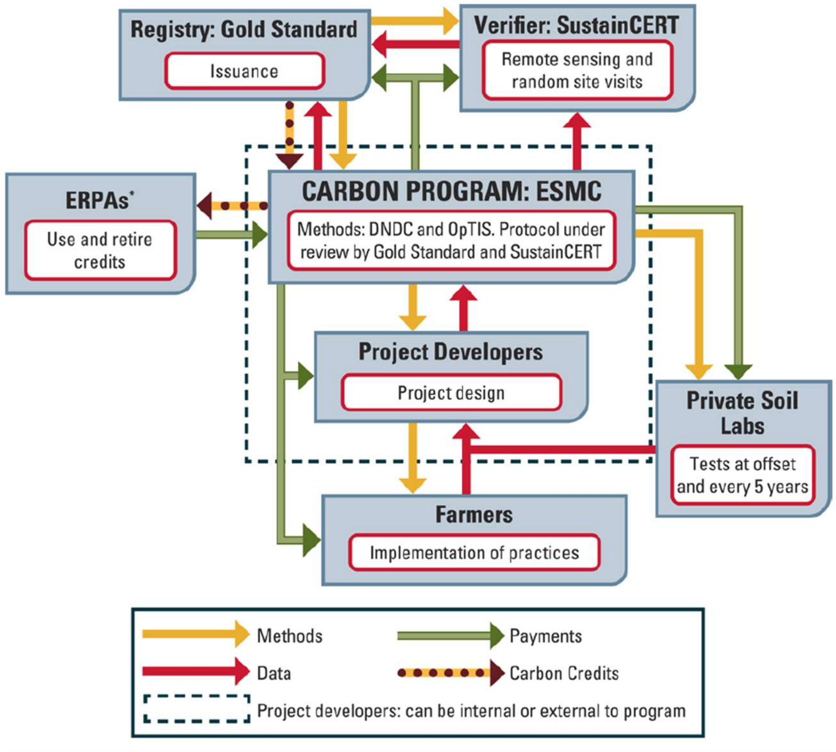


* ERPAs: Emission Reduction Purchase Agreements

Figure 2.5. Carbon credit generation through Corteva Agriscience.

2.4.6 Ecosystem Services Market Consortium (ESMC)

ESMC finds investors to finance projects through ERPAs (Figure 2.6). Its methodology to translate agricultural practices into carbon credits is based on the DeNitrification-DeComposition (DNDC) model and the Operational Tillage Information System (OpTIS) model, which are publicly available. ESMC's methodology is under review by the Gold Standard registry and SustainCERT. Project developers can be internal or external to ESMC. Practices implemented by farmers are independently verified by SustainCERT. Soil tests are mandatory at offset and every five years. The Gold Standard registry issues serial numbers for carbon credits to ESMC, which in turn transfers them to investors. Farm production data is shared with project developers, ESMC, SustainCERT, and the Gold Standard registry. Payments to all actors in the process are distributed over the life of the project. ESMC offers 10-year contracts, and does not require a minimum enrollment acreage.



* ERPAs: Emission Reduction Purchase Agreements

Figure 2.6. Carbon credit generation through ESMC.

2.4.7 Gradable

Gradable is its own registry and marketplace, and it develops its own methodology to translate agricultural practices into carbon credits based on a proprietary model (see Figure 2.7). Project developers can be internal or external to Gradable. Practices implemented by farmers are registered online in the Farmers Business Network (FBN) Platform. Verification relies on remote sensing and is internal to Gradable (2020). Soil tests are required at project offset and possibly later. Gradable issues a serial number for carbon credits generated in a project and assigns 60% of the credits to the farmer, retaining the remaining 40%: 25% of the credits are retained to cover avoidable and unavoidable losses of carbon over a 100-year period and the remaining 15% are retained as fees. Farmers sell their carbon credits through Gradable's online marketplace to end users and brokers (who ultimately resell them to end users), and receive full monetary compensation from which fees to external project developers (if any) are paid. Gradable issues payments to soil labs. Farm production data are shared with project developers and Gradable. Payments start flowing into the system when a sale of (issued) carbon credits occurs. Gradable requires that at least 250 ac be enrolled in the program (although minimum area requirements vary by tier) for five years.

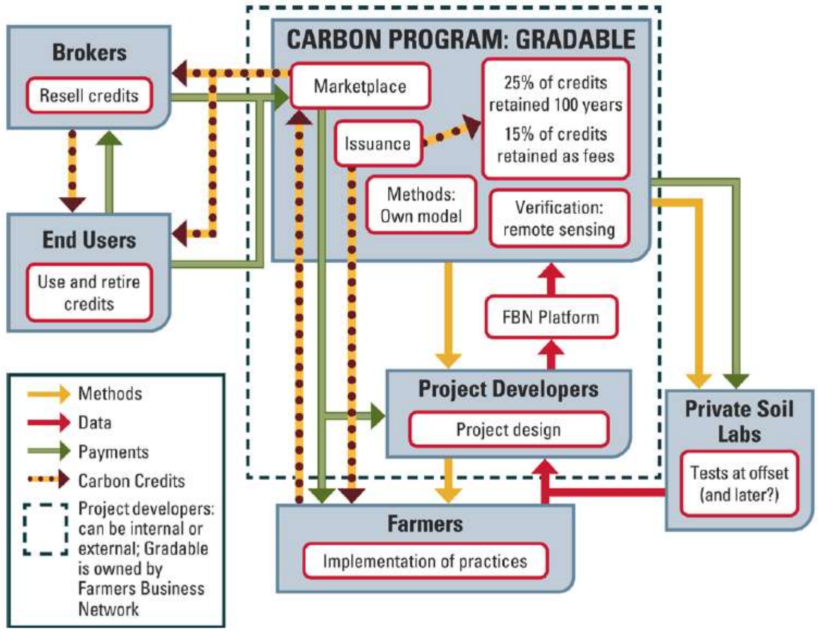


Figure 2.7. Carbon credit generation through Gradable.

2.4.8 Indigo

Indigo develops carbon projects under standards developed by independent, nonprofit standards organizations, with credits issued and tracked on public registries (Figure 2.8). They currently work with the Soil Enrichment Protocol, adopted by the Climate Action Reserve, and the Methodology for Improved Agricultural Land Management (VM0042), coauthored by Indigo and approved by Verra. Indigo works either directly with farmers or through partner organizations (e.g., Corteva) to enroll in the carbon project and adopt new practice changes. Management data collection occurs through a proprietary software platform, as well as through remote sensing and farm management system (i.e., software used by farmers to manage data) integrations. Prior to each issuance by the registry, Indigo hires an independent, accredited verification body that conducts limited site visits and in-depth reviews of all documentation, reporting, and quantification. The program is certified Ag Data Transparent and farm data are not shared beyond the registry and verification body. A portion of credits (5–20%) are permanently held by the registry in a buffer pool to protect against future carbon releases. The balance of credits is issued to Indigo and then either transferred to or retired on behalf of the credit buyers. At least 75% of the proceeds from credit sales are paid directly to farmers. If an unavoidable reversal of stored carbon occurs, the registry uses an equivalent amount of credits from the buffer pool to compensate for the loss. Indigo requires farmers to enroll at least 150 ac for five years to participate in the program.

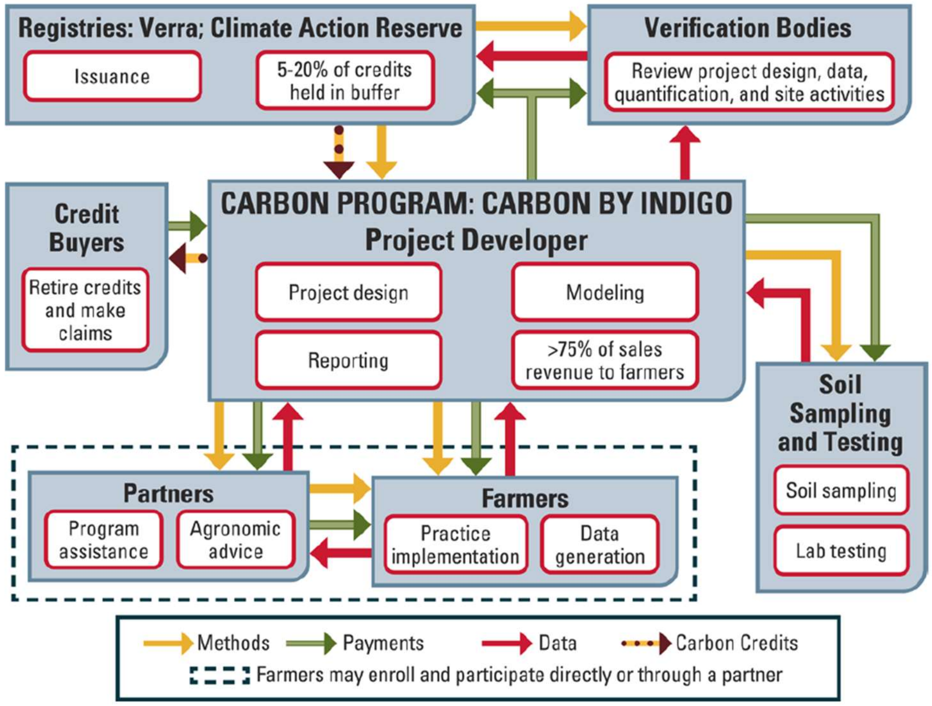


Figure 2.8. Carbon credit generation through Indigo.

2.4.9 Nori

Nori is its own registry and marketplace, and its methodology to translate agricultural practices into carbon credits is based on the publicly available COMET-Farm model (see Figure 2.9). Project developers can be internal or external to Nori. Practices implemented by farmers are verified by independent third parties. Farmers must pay out-of-pocket for the verification process at offset and every three years, and may choose to pay out-of-pocket for soil testing services to ground-truth the estimated carbon sequestration from Nori's model. Nori uses blockchain technology to issue and track serial numbers for carbon credits that are sold to end users and brokers (who ultimately resell them to end-users). Nori adds 15% to the price of carbon credits as fees. After retaining an undisclosed share of the revenue from the sale as a cash reserve to avoid carbon reversals (i.e., disadoption of practices), Nori issues payments to project developers and farmers. If farmers avoid carbon reversals for 10 years following the sale, Nori transfers the retained revenue to them. Farm production data are shared with project developers, independent verifiers, and Nori. Payments start flowing into the system when a sale of (issued) carbon credits occurs. Nori requires farmers to enroll at least 1,000 ac for 10 years (and avoid carbon reversals for another 10 years) to participate in the program.

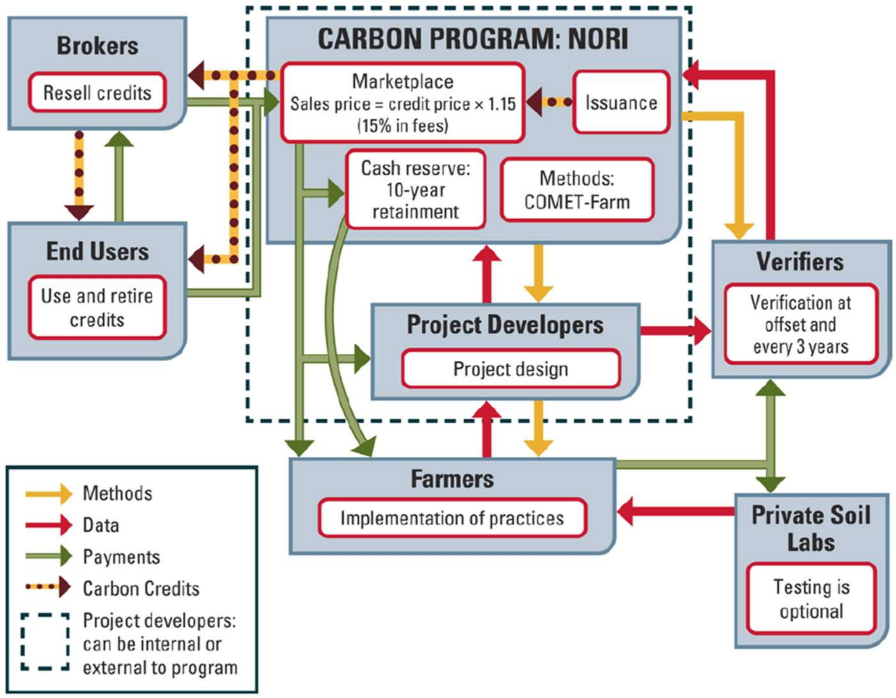


Figure 2.3.9. Carbon credit generation through Nori.

2.4.10 Soil and Water Outcomes Fund (SWOF)

SWOF finds investors to finance projects through ERPAs and acts as its own registry. Its methodology to translate agricultural practices into carbon credits is based on the publicly available COMET-Farm model (Figure 2.10). Project developers can be internal or external to SWOF. Practices implemented by farmers are verified internally by SWOF, and soil tests are mandatory. SWOF issues the serial number for carbon credits generated in a project, transfers ownership of the serial number to the investor, and makes payments to all actors in the process. Farm production data are shared with project developers and collected through an online platform owned by SWOF. Payments are distributed over the life of the project. SWOF offers annual contracts with no minimum enrollment area requirements, and payments are based on implemented practices that remove carbon and provide environmental co-benefits (improvements in water quality, biodiversity, etc.). Current payments range from \$20 to \$45 per ac per year.

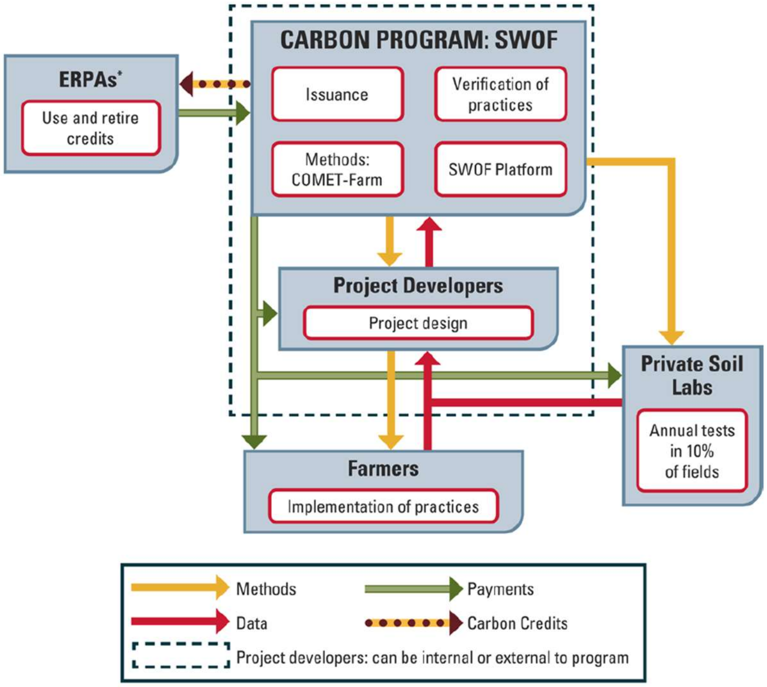


Figure 2.10. Carbon credit generation through Soil and Water Outcomes Fund.

2.5 Overcoming Challenges to Voluntary Agricultural Carbon Markets[6]

As many countries and corporations around the world accelerate GHG emissions reduction and aim for carbon neutrality, demand for carbon credits will likely increase. However, because carbon credits and offsets are credence goods,[7] scaling up voluntary agricultural carbon markets faces multiple challenges, both from the demand and the supply side.

2.5.1 Demand-side Challenges

Issues that can undermine a market for credence goods are well known in economics. Where labels or certification are used to verify a claim on a credence good, markets fail in the presence of difficult to verify claims; a misunderstood or poorly worded label; lack of clear, consistent, and uniform guidelines across certifying parties; lack of trust in certifiers (especially when these are not independent third parties); and label proliferation (the existence of too many labels in a market or on a good leading to confusion about competing claims). Economists have examined these issues in other areas: Giannakas (2002) and Bonroy and Constantatos (2015) examined information asymmetries in the organics markets, concluding that a viable market must have viable certification and undermining of the labels could do great damage to the industry. When Bithas and Latinopoulos (2021) elicited consumers' willingness to pay for carbon sequestration in a stated preference experiment of forest product consumption, they asserted to the respondents that the carbon truly was being sequestered, something that may only be

[6] This section is adapted from Wongpiyabovorn et al. (2021).

[7] Credence goods are goods with qualities that cannot be ascertained by consumers even after consumption (Darby and Karni 1973). A carbon credit or offset based on a claim that GHGs have been removed from the atmosphere or emissions have been avoided through certain processes is a credence good.

inferred in a real market. In the absence of verification, adverse selection (Akerlof 1970) may lead to a market failure of a carbon sequestration claim. As is seen in the variety of third-party certifiers in the carbon sequestration market today, the need for verification is already understood.

Consumers would likely not trust the manufacturer to correctly self-report carbon sequestration because it is arduous for consumers to detect whether a firm's suppliers follow carbon sequestration processes. Certification agents (public or private) who specialize in such detection are necessary in cases where the labels signal the production methods, regional sourcing, environmental impacts, safety, or quality of a good. The absence of the label for a desirable attribute creates a "lemons problem" (Akerlof 1970) where consumers with a higher willingness to pay for a carbon credit cannot detect the attribute in the absence of a label and will not believe it in the absence of certifier credibility. The market will fail not because of a lack of demand but because of a lack of information. Caswell and Mojduszka (1996) and Marette and Roosen (2011) delved into this issue in the case of food labeling; Crespi and Marette (2003, 2005) examined the issue in the case of public labels and eco labels, respectively, while Roe and Sheldon (2007) and Roe et al. (2014) examined the literature on credence good labels in general.

Without government-backed standards, we should expect questionable carbon claims and an increase in competing claims, so-called "label proliferation." Kiesel and Villas-Boas (2013) and Marette (2014) address this issue, which arises when products and markets contain multiple labeled attributes. The concern here is a different type of market failure where consumers become so overwhelmed by competing messages that they lower their willingness to pay for an attribute because of the noise. Label proliferation leads to a "crowding out" of desirable attributes similar to Akerlof's lemons problem. In short, in the absence of standards and verification, buyers of carbon credits and the downstream consumers of credit buyers' products or services may be reticent to assign much value to a GHG sequestration or emission reduction claim.

Another challenge in voluntary carbon markets is that entities promising net-zero emissions or specific GHG emissions targets usually place the target date a decade or more into the future. While such behavior makes sense from a planning perspective, it also allows those entities to commit some investments at the time of the initial announcement, and then postpone further investments until near the target date. The disconnect between long-term voluntary goals and short-term annual purchases of carbon credits or investments in carbon credit generation could result in pent-up demand in years of large announcements, followed by years of low demand and prices, and again high demand in target years. Such cyclicality, combined with the multi-year processes required to produce agricultural credits, could generate incentives to discontinue carbon sinking practices and disrupt the supply of carbon credits prior to the target years.

Although not currently a barrier to the development of agricultural carbon markets, the carbon footprint of the whole system involved in generating carbon credits—including issuance and tracking of the serial numbers for each project in the carbon registries, along with financing projects and trading credits—could become a concern for consumers of carbon credits or the end products or services where carbon credits are applied to reduce their carbon footprint. For example, West and Marland (2002) found that the carbon stored in soil organic matter by reduced-tillage is offset by the GHG emissions into the atmosphere through increased production, transportation, and application of chemicals. Another example is an afforestation program under carbon markets in a specific region that could result in net losses in stored carbon because of the intensification of agricultural production in unregulated regions (Haim et al. 2016). Carbon programs that use energy-intensive accounting and verification systems (e.g., blockchain technology) might generate net-positive carbon emissions and could become less desirable than carbon programs with a smaller GHG footprint.

2.5.2 Supply-side Challenges

Related to the credence attribute of carbon credits, farmers may be reticent to change production practices in order to generate carbon credits of unknown value. Likewise, in the face of an uncertain market, lending institutions may be reticent to fund producers who possibly need specific assets for the production methods applied in the generation of carbon credits.

Accurate measurement and verification of carbon credits from agricultural and forestry activities are typically difficult and costly (van Kooten 2008). Collecting soil samples and measuring soil organic carbon (SOC) is currently the most accurate way to measure the amount of carbon stored in the soil, but it is too costly and time-consuming to be widely used (Castagné et al. 2020). Data collection from satellite mapping may provide an accurate calculation of soil carbon at a lower cost. However, this method is still not able to account for roughness, soil moisture, and vegetation cover, which would lead to less robust estimation (Angelopoulou et al. 2019).

As described in **Section 2.3**, voluntary carbon programs follow different protocols based on different models to calculate how much carbon is removed through the implementation of agricultural practices. The complexity involved in comparing potential carbon credits generated by one specific practice in a particular farm across programs could discourage objective technical comparisons of programs, resulting in farmers choosing programs with the best customer service rather than the highest potential profitability.

Farmers are interested in engaging in carbon markets, but there is uncertainty. In 2021, the Iowa Farm and Rural Life Poll (IFRLP), an annual survey of Iowa farmers (IFRLP 2021), posed a series of statements to measure farmers' attitudes regarding potential carbon programs (see Table 2.1). Farmers were asked to rate their agreement or disagreement with the statements on a five-point scale from "strongly disagree" to "strongly agree." Three items explicitly examined carbon capture. Fifty-two percent of farmers agreed with the statement, "Programs or markets for carbon capture should be developed to help farmers earn money from adoption practices that capture carbon." Similarly, 53% of farmers agreed with the statement, "I would participate in programs that pay farmers to capture carbon through soil health practices," and 47% agreed with the statement, "I am interested in participating in markets or programs that pay farmers to capture carbon in soils through practices such as cover crops and no-till." A related statement that was not specific to carbon capture, "programs and markets that pay price premiums to farmers who document high levels of conservation practice use are a good idea," garnered agreement from 59% of farmers. These results suggest that, in general, Iowa farmers have favorable attitudes toward programs and market mechanisms to incentivize carbon capture. However, because the IFRLP survey is a longitudinal panel survey and not a random sample survey, the results are not necessarily representative of all Iowa farmers. Although medium and large farms, as well as conservation-oriented farmers, might be oversampled in the IFRLP, the survey provides important information about diverse Iowa farmers' perspectives on key agricultural and rural issues.

Table 2.1. Iowa farmers' perspectives on carbon capture initiatives. Data are preliminary from the Iowa Farm and Rural Life Poll 2021 (IFRLP 2021).

	Strongly Disagree	Disagree	Uncertain	Agree	Strongly Agree
	—Percent—				
Programs or markets for carbon capture should be developed to help farmers earn money from adopting practices that capture carbon.	4.8	7.1	36.2	41.8	10.1
I would participate in programs that pay farmers to capture carbon through soil health practices.	3.4	6.5	37.1	37.8	15.1
I am interested in participating in markets or programs that pay farmers to capture carbon in soils through practices such as cover crops and no-till.	5.5	8.7	38.5	37.7	9.7
Programs and markets that pay price premiums to farmers who document high levels of conservation practice use are a good idea.	2.6	6.2	32.7	47.5	11.0

Non-additionality is one of the major risks making conservation programs cost-ineffective. Agricultural conservation practices yield additional environmental gain only if they would not have been adopted without payment. Estimating additionality for selected agricultural practices, Claassen et al. (2018) conclude that the adoption of three off-field structural practices (filter strips, riparian buffers, and field borders) and the elimination of fall application of nitrogen fertilizer were highly additional, while the adoption of conservation tillage was only moderately additional. Sawadgo and Plastina (2021) estimate that cover crops were moderately additional and that more than half of the cover cropped area funded through cost-share- programs would not have been planted without the cost-share payments. The 11 voluntary agriculture carbon credit programs analyzed by Plastina and Wongpiyabovorn (2021) require additionality to generate a carbon credit. However, not all programs require that farmers change their production practices because programs use a wide array of benchmarks to determine what is additional or different: some programs require a change of practices with respect to past practices on the same field, while others require that practices in the field be different from common practices in the area (even if the same practices have been implemented for many years in the field under consideration).

Permanence is another major driver of carbon credit quality. Generating high-quality credits with long-lived carbon storage in the soil is a costly process because of the required changes in farming practices that sometimes reduce productivity—even if temporarily—and the costs to verify and certify the carbon sequestration. For example, no-till could reduce crop productivity, particularly in cooler and/or wetter climatic conditions because of the surface residues and lower soil temperatures (Ogle et al. 2012). According to Gramig and Widmar (2018), farmers in Indiana who have never adopted any conservation tillage or no-till would require almost a $40 per ac increase in net revenue to implement no-tillage, while individuals who previously used conservation tillage would be willing to adopt without payment. They also found that an additional $10.57 per ac is needed to enter the program with a multi-year contract that does not allow them to change their tillage practices during the contract term. Having a carbon project certified to generate high-quality carbon credits according to the Gold Standard registry can cost $5,000 in one-time validation fees and $3,500 per year in annual verification and registry fees (Gold Standard 2021). Furthermore, Plastina and Wongpiyabovorn (2021) report that when contracted practices are temporarily discontinued because of factors external to the farm (e.g., weather), some voluntary agriculture carbon programs impose penalties associated with skipping payments for the discontinued practices until reinstated (Soil and Water Outcomes Fund, CIBO Impact) or until additional gains in carbon sequestration are observed (ESMC, Indigo), and at least two initiatives do not have any penalties for permanent dis-adoption (Gradable, Bayer).

In the present environment of burgeoning agricultural carbon programs, little attention is paid to the potential effects of alternating adoption, opportunistic adoption, and partial adoption on total area under conservation practices (Pannel and Claasen 2020), let alone their limiting effects on the development of voluntary carbon markets. Carbon reversal from dis-adoption of conservation practices can occur when a participant of a carbon program stops using the contracted practice when the contract expires. Jackson-Smith et al. (2010) studied a single watershed in Utah between 1992 and 2006 and found that 66% of crop production practices implemented were still maintained in 2007, and 32% of the practices that were discontinued were driven by farmers exiting farming or selling land for non-farm development. Using county-level data from the 2012 and 2017 United States Censuses of Agriculture, Sawadgo and Plastina (forthcoming) evaluate regional patterns of adoption and disadoption of conservation practices in the United States. They estimate that national disadoption rates in cover crops and no-till averaged 15.60% and 39.38%, respectively, between censuses. Plastina and Sawadgo (2021) report that 11% and 33% of the counties in Iowa, Illinois, and Indiana disadopted cover crops and no-till, respectively, reducing their areas in those conservation practices by 25% and 13% between 2012 and 2017. If these percentages are indicative of the probability that farmers participating in voluntary carbon programs could temporarily discontinue contracted practices and trigger penalties from carbon programs, those findings suggest that farmers planting cover crops and using no-till would face non-trivial probabilities of being penalized over the life of a multi-year carbon contract.

Even within a credible verification and certification system mitigating uncertainty in the conversion of agricultural practices into carbon credits, suppliers of agricultural carbon credits will face competition from other suppliers of carbon credits generated in forestry, geological carbon sequestration, ethanol production with carbon capture and sequestration, landfill methane capture and destruction, and multiple other sources. The quality of credible agricultural carbon credits, dependent mostly on the degree of additionality and permanence of the carbon sequestration, will play a critical role in the determination of payments received by farmers (via direct sale of credits to end users and brokers, or indirectly via carbon programs that sell credits to investors).

The cyclicality in demand for carbon credits resulting from strategic behavior by entities with voluntary GHG emissions targets could, as explained above, generate an oversupply of credits and a decline in credit prices in the middle of the cycle.

Although outside the context of carbon programs, multiple studies have examined barriers to adoption of conservation practices, suggesting that a diverse combination of economic and agronomic factors, social norms, perceptions of government programs, farm characteristics, land tenure factors, and knowledge-related factors can pose barriers to conservation adoption (Nowatzke and Arbuckle 2018; Prokopy et al. 2008, 2019; Ranjan et al. 2019).

A further barrier to participation in carbon programs is lack of transparency in the price discovery mechanism for participating farmers. Farmers and ranchers interested in carbon programs are currently being offered anywhere between $10 to $40 per ac to implement practices that will generate carbon credits, but prices will be subject to market fluctuations beyond pilot programs (Plastina and Wongpiyabovorn 2021). In March 2020, the CME Group began trading CBL Global Emission Offset (GEO) futures contracts. The aim of these futures contracts is to help manage risk in carbon prices and establish a global pricing benchmark for the voluntary emissions offset market (CME Group 2021a). In August 2021, the CME Group also started trading futures contracts for offsets generated from agriculture, forestry, and other land use, called Nature-Based GEO (N-GEO). To ensure the transparency of N-GEO futures, only the offsets from Verra's Verified Carbon Standard for Agriculture, Forestry, and Other Land Use projects and/or the Climate, Community, and Biodiversity Standards are accepted for trading (CME Group 2021b). As of August 20, 2021, the average prices of GEO and N-GEO futures were $5.11 and $7 per Mt CO_2e, respectively. Trading volumes in August 2021 averaged 198 and 503 contracts per day (equivalent to 0.2 and 0.5 million Mt CO_2e) for GEO and N-GEO futures, respectively, with open interest of 835 and 6,092 contracts at the end of the month. The lack of "hard" caps on GHG emissions in voluntary programs and the small number of carbon credits traded, together with the cyclical pattern of demand for carbon credits, and the resulting lack of volatility to attract speculators that inject liquidity in the market are major reasons to be skeptical about the ability of GEO and N-GEO futures to serve as a pricing benchmark for voluntary agricultural offsets (Wongpiyabovorn et al. 2021).

Conservation practices can not only remove carbon and reduce GHG emissions, but they can also benefit farmers by reducing soil erosion, improving water infiltration, soil water storage, and soil quality. In addition, cover crops and proper nutrient management could improve water quality by reducing nitrate leaching and phosphorous runoff to nearby water bodies. However, the co-benefits from adopting these practices are uncertain and take time to develop. For example, the adoption of no-till/strip-till takes more than five years to yield reduced soil erosion and sediment loss to water and wind and an increase in water-storage capacity (Toliver et al. 2012). If policy-makers choose to incentivize farmers' participation in carbon and ecosystem services programs through subsidies or cost-share programs, it is important to keep in mind that uniform payments across geography and/or based on adopted practices are not cost-effective to deliver desirable environmental outcomes (Khanna 2017). Secchi and Jones (2021) propose that government subsidies be used to support long-term or permanent practices such as land retirement and reforestation because of their associated water quality and habitat co-benefits, rather than investing in carbon capture and storage projects at ethanol plants.

Finally, as long as buyers of agriculture carbon credits perceive differences in the quality of credits generated through alternative protocols, it can also be expected that some programs will gain market share and some will exit the market, affecting systemic risks for farmers and credit buyers (Plastina and Wongpiyabovorn 2021). The risk to farmers could be partially mitigated through the standardization of equivalences for carbon farming practices across initiatives and the introduction of transferable partial and full credits across protocols. However, the risk of a shorter than expected permanency of a carbon credit triggered in the event that a program exits the market and farmers who sold credits through that program discontinue the practices before the expiration of the retention period is only partially mitigated in a few programs through retained carbon credits. Credit reversals are a liability for which there is no insurance policy currently available.

2.5.3 A Way Forward for Voluntary Agricultural Carbon Programs

A textbook example of overcoming a market failure for credence goods is the case of United States organic markets before and after certification. Prior to specific standards for the production, the market for organics was very small with lenders reluctant to finance operations. Once standards were set and claims were verified, many farmers overcame their reluctance to join the industry, consumers overcame their distrust of product claims, and lenders had a greater understanding of the needs of producers in this new market (Giannakas 2002; Klonsky and Smith 2002; Kostandini et al. 2011; Jones et al. 2015).

With the 2021 COP26 United Nations (UN) Climate Conference in Glasgow formally agreeing to set standards on carbon credits worldwide (Article 6), the markets for such credits are poised to take on new levels of interest (UN Climate Change 2021). If nations are now mostly in agreement on the units of measurement and standards for the counting of carbon mitigation credits—a goal that eluded the parties to the 2015 Paris Agreement—carbon credit markets may overcome much of their previous participation disincentives though clearly risks remain.

A major piece of legislation in support of increasing transparency and standardization in voluntary agricultural carbon programs is the Climate Solutions Act of 2021 (GCSA), passed by the United States Senate on June 24, 2021. Once ratified by the United States House of Representatives, the GCSA will assist farmers, ranchers, and private forest landowners with participation in voluntary carbon markets and adoption of conservation practices. Particularly, the legislation will provide the United States Department of Agriculture (USDA) authority to create a GHG Technical Assistance Provider and a Third-party Verifier Certification Program. Although the bill does not specify any details about carbon markets, it instructs the Secretary of Agriculture to provide necessary definitions of the markets and determine the rules for the certification program (Crespi and Tidgren 2021). An effort to standardize or create equivalencies to the amount of carbon credit generated by the same practice in the same farm across private programs would add transparency and reduce systemic risks for potential participants.

An international survey conducted by the United Nations Development Programme (UNDP) and the University of Oxford found that 64% of respondents agree that climate change is a global emergency and call for broad climate policies, such as more renewable energy, adopting climate-friendly farming practices, and conserving forests and land (UNDP 2021). Likewise, about 65% of surveyed Americans desired the federal government to take more action on climate change (Tyson and Kennedy 2020). However, the implementation of climate policy has encountered multiple challenges, in part because of less than full agreement on the science of climate change. Additionally, the disbelief of a substantial share of representatives in the United States Congress about the science of climate change slows environmental policy discussions. Drennen and Hardin (2021) reported that 26% of elected officials in the 117th Congress reject the evidence of human contribution to climate change and support the continued usage fossil fuels.

2.6 CO_2e Removal Potential by Tier of Carbon Prices

The most recent assessment of negative emissions technologies by the National Academy of Sciences, Engineering, and Medicine (NASEM 2019) evaluates six major technical approaches to carbon removal, two of which are closely related to agricultural production. The first approach, terrestrial carbon removal, relies on three negative emissions technologies: land use and management practices such as afforestation/reforestation, changes in forest management, and changes in agricultural practices that enhance soil carbon storage. The second approach, bioenergy with carbon capture and sequestration (BECCS), relies on one negative emissions technology: using plant biomass to produce electricity, liquid fuels, and/or heat combined with capture and sequestration of any carbon dioxide produced when using the bioenergy and any remaining biomass carbon that is not in the liquid fuels (see **Chapter 5** in this report for more details). The NASEM indicates that these four negative emissions technologies are ready for large-scale deployment with low to medium costs ($100 per Mt CO_2 or less) and substantial potential for safe scale-up from current deployment.

The NASEM report identifies the maximum potential rates of carbon reduction and removal that could be achieved safely and economically, given the current knowledge and level of technological development. "Safe" means that the deployment would, with high confidence, not cause large potential adverse societal, economic, or environmental impacts. "Economic" means that the deployment would cost less than $100 per Mt CO_2e, approximately equal to $1 per gallon of gasoline, because combustion of a gallon of gasoline releases approximately 10 kg of carbon dioxide. Overall, the price for carbon is expected to have a substantial impact on the amount removed (Figure 2.11).

2.6.1 Afforestation/Reforestation

Planting trees or facilitating natural regeneration of trees on land that has been in a non-forest use condition for some time have the potential to remove up to 150 million Mt CO_2e per year in the United States, at a cost of up to $20 per Mt CO_2e (NASEM 2019). The estimation assumes full adoption of forestry management practices described in the NASEM report.

Two major limiting factors to afforestation/reforestation are the availability of land given the population's needs for food and fiber production and for biodiversity, and the inability to fully implement all forestry management practices.

2.6.2 Changes in Forest Management

Various forest management activities are expected to remove carbon from the atmosphere, including accelerating forest regeneration in areas that suffered major disturbances; restoring forests that have been converted to unsustainable forest conditions; extending harvest rotations to grow larger trees and sustain carbon removal rates; maintaining healthy forests by treating areas affected by insects and diseases or preventing conditions that foster outbreaks; and thinning and other silvicultural treatments that promote overall higher stand growth compared with untreated conditions (NASEM 2019).

The NASEM report estimates that under full adoption of changes in forest management, the United States has the potential to remove up to 100 million Mt CO_2e per year through changes in biomass and soil carbon,[8] at a cost of up to $20 per Mt CO_2e (NASEM 2019). Two major limitations to achieving the potential removal rate are the demand for wood and the inability to fully implement all forestry management practices.

[8] This estimate excludes changes in the stock of harvested wood products, or emissions reductions from increased use of harvested wood in place of other materials.

2.6.3 Agricultural Practices to Enhance Soil Carbon Storage

This technology includes increased productivity and residue retention, cover crops, no-tillage and other conservation tillage, manure and compost addition, conversion to perennial grasses and legumes, agroforestry, rewetting organic (i.e., peat and muck) soils, and improved grazing land management (NASEM 2019). The NASEM report estimates that under full adoption of agricultural practices to enhance soil carbon storage, the United States has the potential to remove up to 250 million Mt CO_2e per year, at a cost of up to $100 per Mt CO_2e (NASEM 2019). Two major challenges to achieving the potential removal rate are the limited rates of carbon uptake by existing agricultural practices on a per acre basis and the inability to fully implement soil conservation practices.

2.6.4 Bioenergy with Carbon Capture and Sequestration (BECCS)

While BECCS typically refers to the integration of trees and crops that extract carbon dioxide from the atmosphere as they grow, the use of this biomass in power plants, and the application of carbon capture and sequestration via carbon dioxide injection into geological formations, the NASEM report also includes biomass thermochemical conversion to fuel with biochar soil amendment, and biomass fermentation to fuel with carbon capture and sequestration. The NASEM report estimates that under full adoption of BECCS, the United States has the potential to remove up to 500 million Mt CO_2e per year, at a cost of $20 to $100 per Mt CO_2e (NASEM 2019). The major limitations to achieving the potential removal rate are the cost of the BECCS technologies, the availability of biomass given needs for food and fiber production and for biodiversity, the inability to fully capture waste biomass, and the current state of fundamental understanding of those technologies (NASEM 2019).

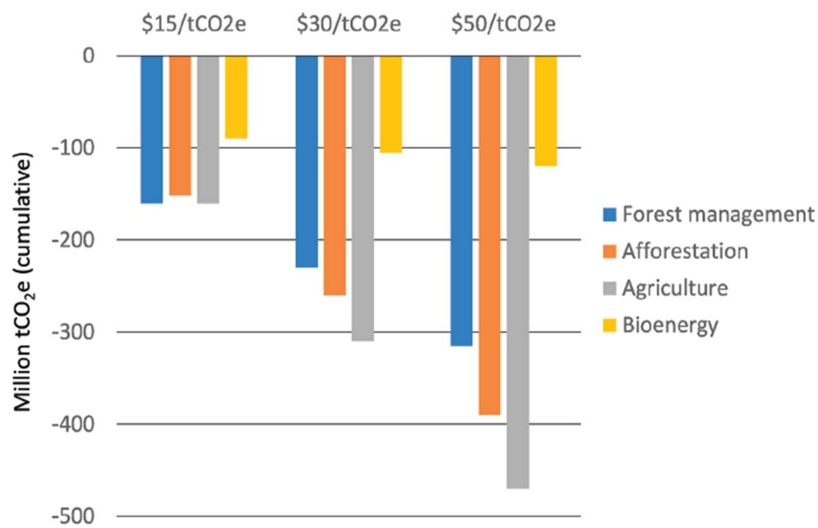


Figure 2.11. Mitigation potential land-based negative emissions technologies in the United States for three carbon price scenarios ($15, $30, and $50 per Mt CO_2e). Negative numbers indicate carbon removal from the atmosphere. Estimates from the FASOM model reported by Baker et al. (2010) and Jackson and Baker (2010). Source: NASEM 2019.

2.7 Research Needs

Following COP 26, developed and developing countries are considering major changes to agricultural policies in order to reduce global GHG emissions. These policies might involve changes in agricultural productivity through reduced fertilizer use or reduced stocking density on pasture, as well as changes in the marginal cost of carbon via domestic or border carbon taxes, or emissions trading systems. The new policies will likely change the international competitiveness of agricultural commodities produced in the United States and, in turn, lead to changes in trade patterns and international production.

In order to inform policy makers about the system-wide impacts of changes in domestic and international agricultural carbon policies, the following high-priority research areas have been identified:

Farm-level effects:

- Elicit the potential extent and intensity of farmers'/ranchers' participation in voluntary carbon programs under alternative payment levels, timing of payments, required practices, stacking of payments from different sources, and contract length. Estimate differences in willingness to accept (WTA) payment for participation in carbon programs by farm size, location, production mix, land tenure (owner-operator vs. tenant) and history of management practices, using a discrete choice experiment through a mixed-mode online-mail survey.
- Evaluate the potential role of landowners in scaling up participation in voluntary carbon programs through soft nudges, lower cash rents, longer contracts, or simply demanding participation, via an additional special section on carbon programs to the statistically representative 2022 Iowa Farmland Ownership and Tenure Survey.
- Develop risk management tools for farmers and ranchers willing to participate in voluntary carbon programs to help them transition into carbon markets, to incentivize them to maintain conservation practices in place after the expiration of the carbon contracts, to integrate carbon and environmental services markets, and to enhance markets for low-carbon ag products.
- Develop case studies for farms in different crop reporting districts with different production mixes, soil profiles, and historical management practices and use these to evaluate the most profitable changes in practices to participate in carbon and environmental markets.

State-level effects:

- Develop a set of plausible scenarios reflecting alternative degrees of farmer participation in carbon programs, adoption of conservation practices and resulting land use change, and corresponding changes in yields and carbon credits generated. These results, in turn, could be used as inputs in economic models such as those developed by Iowa State University's Center for Agricultural and Rural Development (CARD).

National and global effects:

- Identify the most likely changes in international policy to be implemented over the next decade, develop plausible scenarios in land use change under the new policies, and use economic models, such as those developed by CARD, to evaluate impacts on agricultural production and trade. Simulate regional changes in land use, commodity prices, and net farm income.

- Assess the changes in global carbon emissions from agriculture under the new international policies by comparing the carbon intensity scores from the International Panel on Climate Change (IPCC) assigned to each commodity and country in the status quo and after the implementation of the new policies.[9]

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[9] As an example of the importance of assessing the global impact of policy changes, an earlier version the CARD Land Use Model was used to evaluate the impact of a tax on United States beef production. The analysis showed that a reduction in United States beef production would lead to an expansion in beef production in Brazil and in other countries that have higher carbon intensity than the United States (Dumortier et al. 2012). The net impact was an increase in global carbon emissions.

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Chapter 3. Measurement, Reporting, and Verification

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3.1 Highlights

- Measurement, reporting, and verification (MRV) for carbon markets is complex and changing: there is great need for standardization.
- Lower-cost, accurate tools for measuring soil organic carbon (SOC) are urgently needed; new tools are emerging that may play a role.
- Further automation of practice data collection from farm machinery to support MRV and practice adoption is needed and likely, but manufacturers need clear market signals on needed data types and formats.
- MRV for reduced methane emissions from manure storage or methane capture is relatively well established and favors high-value credits and/or potential useful, marketable product (e.g., renewable natural gas).
- There is a clear need for close collaboration of agronomists/soil scientists, statisticians, data management experts, farmers, agriculture retail, carbon market brokers, and others to design and help standardize efficient and accurate (and therefore credible) ways to gather required data.

3.2 Background

Agricultural practices can be used to remove carbon dioxide from the atmosphere and generate carbon credits in an effort to reduce greenhouse gas (GHG) emissions related to food production. Agricultural carbon credits can be used as offsets in compliance and voluntary markets. While only a few agricultural practices are listed among the accepted sources of carbon credits in United States compliance markets, the potential demand for agricultural carbon credits in voluntary markets seems to be sizable. In anticipation of a rapid growth in demand for carbon credits by the voluntary market, multiple carbon programs are competing to attract the attention and commitment of farmers in the United States. The attributes that carbon programs use for marketing include length of contract, payment per practice versus payment per results, look-back period (i.e., payment for practices that have already been implemented), dedicated agronomic and customer service support, and payment in cash versus payment in carbon credits – and for those who pay with carbon credits: immediate sale versus allowing farmers to choose the timing of credit sale. Marketing strategies towards farmers seldom mention the methodology that will be used to translate changes in agricultural practices into carbon credits. In that sense, carbon programs can be considered “black boxes” that transform changes in practices into carbon credits.

Carbon programs use very different marketing strategies to attract the attention of potential buyers of carbon credits. Buyers are typically interested in high quality, or credible, carbon credits and prefer to avoid buying "lemons" or high-price and lower credibility credits because that would reflect poorly in their annual environmental, social, and governance (ESG) reports. If credits are maintained as assets in their balance sheets, they could generate net worth losses under more stringent intangible valuation rules currently under consideration by the United States Securities and Exchange Commission. A key attribute of carbon programs considered by buyers of carbon credits is the measurement, reporting, and verification (MRV) system in place to provide assurance that an unobservable good (i.e., the reduction or removal of a metric ton of CO_2e) has been produced according to certain protocols.

MRV systems are costly to implement given that all carbon programs are nascent (e.g., in 2021 Nori celebrated the sale of its 50,000th ton of CO_2e, four years after starting its operations). The cost to operate current MRV systems is financed by venture capital or absorbed as start-up costs. Carbon programs are testing multiple approaches to MRV systems at high costs in the understanding that they will learn by doing and perfect techniques to reduce costs to a bare minimum after venture capital runs out, so they can become financially viable. *The survival of carbon programs relies on the expectation that better MRV systems will become available at lower costs.* During this initial period of frequent trial and error, carbon programs are expected to change registries, protocols, project design methods, carbon estimation models, and verification strategies multiple times before settling for more stable rules. Clear, overarching rules and guidance, which are expected to come out of the Growing Climate Solutions Act of 2021, could shorten the trial and error period and contribute to an earlier gelling of an agricultural carbon market.

In the understanding that MRV systems constitute key elements of the commercial strategies of carbon programs and that they are currently in a fluid state, this chapter discusses selected MRV practices based on publicly available information to provide a general understanding of recent developments. However, we expect these MRV systems to continue to evolve rapidly and become even more complex and diverse before settling into a coherent set of rules. This chapter primarily addresses the measurement component of MRV, which is often referred to as "monitoring."

3.3 Measurement, Reporting, and Verification

Carbon markets depend on MRV protocols to assure that purchased credits can be associated with specific practices being implemented and maintained in specific locations. This process requires tradeoff between certainty of the carbon credit and the cost of the protocol, with greater certainty achieved at higher costs. Key terms are defined as follows:

- **Measurement, Reporting, and Verification (MRV):** "A system or protocol for tracking specific methods or outcomes, transparently communicating specific information, and validating that the information is accurate and complete" (Oldfield et al. 2021).
- **Measurement:** Quantification of the sources and removals of GHGs. These sources and removals include emissions, emissions reductions, and/or removal of carbon dioxide from the atmosphere (UNFCCC 2014). More specifically at the farm level, measuring involves estimation of carbon credits based on farm management data combined with field-based sampling,

modeling, a combination of sampling and modeling, or sampling combined with remotely sensed data. The credits are recorded by a carbon registry.

- **Reporting:** Conveyance of information on performance under carbon programs.
- **Verification:** "The process by which an accredited third-party verifier examines or reviews a project, including the methodology and emission reduction or removal calculations, that the regenerative practices are occurring on the farm and that SOC is being properly accounted for" (Oldfield et al. 2021). Programs focused on methane typically have similar third-party verification processes.

Reducing the cost of MRV can facilitate the expansion of carbon market opportunities. Purchasers of credits, brokers, farmers, and others directly involved in the carbon markets rely on these data and processes. Government entities, businesses, and other parties not directly involved in carbon market transactions are also seeking the best possible information to set policies and position for economic development. In addition to the cost of collecting data, there are real costs associated with managing the data.

Approaches to carbon program data collection at the farm level for initial planning and MRV vary widely. Potential increases in cost effectiveness and accuracy of MRV need to be considered within the context of current approaches. These data needs include the location and extent of management practices that remove or reduce carbon and make changes in soil carbon content and greenhouse gas emissions. Data on soil erosion may also be relevant for understanding changes in soil carbon at the field level, as soil carbon may be translocated within fields as a result of erosion.

The initial planning phase for implementing carbon credits typically involves some combination of a site visit, soil sampling, remote sensing, and in some cases a "desktop review" using existing data. The desktop review element incorporates collection and analysis of data from farm management tools and decision support tools that estimate carbon credit generation. These services are often provided by agriculture retailers, carbon programs, or partnerships thereof.

There are numerous potential changes in policy and legislation that are likely to impact MRV, including agreements at the recent United Nations COP26 in Glasgow, the Growing Climate Solutions Act of 2021 currently pending in the United States Congress, and initiatives by the White House. As noted in **Chapter 2**, new rules for carbon markets, such as how to count carbon offsets, were established at COP26 (COP26 2021; Krukowska 2021). In general, there will be greater constraints and increased consistency in carbon markets worldwide. This will likely influence the many details of MRV of carbon offsets, but the specific details will be developed in a series of meetings extending through much of 2022. The Growing Climate Solutions Act of 2021 is a major piece of legislation passed by the United States Senate in June 2021 that has the goal of increasing transparency in, and standardization of, voluntary agricultural carbon programs. It is expected to assist participation of farmers, ranchers, and private forest landowners in voluntary carbon markets (Crespi and Tidgren 2021). The White House (2021) recently released a United States Methane Emissions Reduction Plan, which includes recommendations for agriculture that may impact MRV for methane projects. These new and potential rules and policies could dramatically change the landscape of MRV.

3.4 Current Approaches to Measurement, Reporting, and Verification

This section provides an overview of current approaches to MRV by both carbon programs that interface with farmers and the various carbon program's registry protocols. These current approaches serve as a reference point for potential innovations.

Carbon programs apply a variety of approaches (see Table 3.1). In addition to differences in the registry and therefore MRV protocol they follow, they use different approaches to third-party verification. The third party in this case is not buying or selling credits, but is an independent entity accredited or certified to provide independent confirmation that program requirements are met, and thereby increase the credibility of the credits. Most programs require some level of soil sampling to establish a baseline ("at offset") and regular testing thereafter at five-year intervals. Informal discussions with those implementing programs indicate a considerable amount of uncertainty, trial and error, and the need for guidance on the details of soil sampling. Additional information on soil testing for soil organic carbon is provided in **Section 3.5.3.1**.

As noted previously, carbon programs are continuing to change rapidly, with frequent entry of new players in the marketplace. This rate of change and diversity of approaches is likely to continue for some time. MRV represents a large cost to these programs, and in general they are very interested in innovations that represent some combination of increased accuracy (credibility = value) or reduced cost. The appetite for innovation is also driven by the highly competitive nature of the current carbon market. Potential innovations include rapid, field-based measurements of SOC (**Section 3.5.3.1**), greater use of farm machinery-based data collection (**Section 3.5.2**), remote sensing (**Section 3.5.4**), and improved models and calculators (**Chapters 8, 9**).

Table 3.1. Summary of monitoring approaches for voluntary programs, adapted from Plastina (2021).

Program	Registries	Third Party Verification	Soil Testing
ESMC	Gold Standard	SustainCERT verifies farmer practices (remote sensing and random site visits)	Soil tests at offset and every 5 years
SWOF	Acts as its own registry	Practices verified internally	Initial, then every five years. Annually tests 10% of fields. Measure TOC and bulk density. Sampling and analysis <u>not</u> based on any outside registry. Site visits and remote sensing used on all fields.
Indigo	Verra and Climate Action Reserve	Practices implemented verified by independent third parties through random site visits and evidence checks	Following Soil Enrichment Protocol (CAR) and VM0042 protocol (Verra). Soil tests mandatory.

Nori	Acts as its own registry and marketplace	Practices implemented verified by independent third-parties at offset and every three years.	Farmers may choose to pay out-of-pocket for soil testing services to ground-truth the estimated carbon sequestration from Nori's model. Testing is optional.
Corteva	Partners with ESMC and Indigo	Practices verified by independent third parties	Mandatory at offset and every five years
Agoro Carbon Alliance	Gold Standard and Verra	Practices verified by independent third parties	Mandatory at offset, and every five years
Bayer Carbon	Registry agnostic: ability to partner with all registries	Protocol-specific verification practices	Soil testing paid by Bayer

An overview of the MRV protocols for common registries is provided in Table 3.2. Although there are similarities, there are clear differences in approach. In general, if a model (or calculator) is used, it is either specified or must be peer reviewed. Most require some mixture of modeling and soil sampling. Similar to the carbon programs in Table 3.1, the variation in approach between the registries suggests there is room for both innovation and standardization.

Table 3.2. Highlights of MRV protocols for selected registries (adapted from Oldfield et al. 2021).

Protocol	Model Required	Baseline Validation (Dynamic vs. Static)	Approach (model, sample, hybrid)	Stratification	Minimum Samples per Strata	Sampling Frequency
Climate Action Reserve Soil Enrichment Protocol v 1.0	No, but model must meet minimum requirements (publicly available, peer-reviewed)	Dynamic performance baseline calibrated with sampling	Hybrid	Required	Three	Every five years
Verra CM0021 Soil Carbon Quantification Method. V 1.0	Yes, DNDC	Static established by sampling	Hybrid	Required	Not specified	At least every five years
NORI Croplands Methodology v 1.1	GGIT	Dynamic performance baseline	Model	Not specified	N/A	N/A

Gold Std. Soil Organic Carbon Framework Method. V 1.0	No, but must be peer-reviewed	Either performance or static depending on accounting approach	Sampling or hybrid	Yes	Not specified	Every five years
Verra VM0042 Methodology for Improved Agricultural Land Management, v1.0 - Verra	No, but model must meet mininum requirements (publicly available, peer-reviewed)	Dynamic performance baseline calibrated with sampling	Hybrid	Recommended	Not specified	Every five years

The following sections provide additional details on MRV for a few example registries as an illustration of the types of data they require. Inclusion of these examples as well as the relative level of detail does not imply endorsement (or lack of endorsement of registries not listed).

3.4.1 Gold Standard Soil Organic Carbon Framework Methodology

The Gold Standard Registry maintains a Soil Organic Carbon Framework Methodology (Gold Standard 2020a), which provides a common basis and approach for several different types of practice protocols (SOC "activity modules"). These activity modules include improved tillage practices, improved cropland management, and other practices that increase SOC from baseline to a future project scenario. Projects may access one or more activity modules. The methodology sets standards that carbon project developers can use and reference when developing and ultimately marketing carbon credits. Gold Standard maintains a registry or formalized database of credits and programs aligned with its protocols. It provides three alternate approaches for quantifying SOC for baseline and "project scenario" (what will exist once the project is implemented and operational) and a general assessment of the accuracy of the approach and the associated likelihood of deductions in the value of credits (Table 3.3).

Table 3.3. Summary of three alternate approaches in the Gold Standard Soil Organic Carbon Framework Methodology. Definitions: SOC = soil organic carbon.

Approach	Gold Standard's Summary Assessment of Approach
Approach 1: Requires onsite (soil) measurements to directly document baseline and project SOC stocks.	"High accuracy, no deductions" (i.e., deduction refers to the likelihood of a reduction in credit value due to the selected approach)
Approach 2: Uses calculation approaches, datasets, parameters and/or models from peer-reviewed publications to estimate baseline and project SOC stocks. Project owners need to prove that the research results are conservative and applicable to the project site and management practice.	"Medium accuracy, deductions possible"

Approach 3: Applies default factors to estimate SOC changes, relating to the general Tier 1or Tier 2 model described in the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC 2019).[10] If possible, the Tier 2 approach as outlined in the IPCC Guidelines should be applied. Applicability of SOC reference values (SOCREF) to be used in connection with IPCC impact factors shall be transparently demonstrated for the project area.	"Low accuracy, deductions likely"

Other specific elements of the Gold Standard SOC protocol include:

- Methane and nitrous oxide: Measurement is focused primarily on carbon dioxide, but methane or nitrous oxide monitoring may be required under some activity modules, reflecting the concerns over possible leakage or the impact of fertilization changes.
- Drainage: The protocol *prohibits* changes in water regimes including drainage.
- Yield: The protocol *prohibits* projects that result in reductions in crop yield, reflecting food security concerns.
- Laboratory analysis: There is no reference to a specific SOC analytical protocol.
- Stratification: The protocols require stratification of both baseline and project scenarios based on:
    - Soil type
    - Climate zone
    - Land management / cropping system
    - Input levels (e.g., fertilization)
    - As applicable (defined in SOC Activity Modules, e.g., Gold Standard (2020b)):
        - Tillage practices
        - Soil properties (e.g., nutrient status or soil health)
        - Hydrology
        - Risk of carbon loss (e.g., fire risk)
    - For each stratum, SOC measurements have to be performed (Approach 1) and/or model parameters identified and verified (Approach 2 or 3).
- Validation: At the time of validation, the project owners are required to document the applicability of parameters and models used in Approach 2 or Approach 3. For each stratum, excavation of small soil test pits is required (~20 inches x 20 inches x 20 inches).

[10] Tier 1 is the least rigorous approach defined by the IPCC. Tier 2 and Tier 3 represent increasing degrees of rigor (and cost) in determining GHG inventories.

3.4.2 Verified Carbon Standard (Verra) – VCS Module VMD0021

The VMD0021 protocol (Verra 2012) includes a number of detailed requirements. Reportedly, no one has adopted this protocol since it was published in 2012 because of the intensive sampling requirements (Oldfield et al. 2021). (As noted in Table 3.1, Verra's VM0042 methodology has been adopted by some carbon programs). All laboratory methods must follow NRCS Soil Survey Manual, Soil Survey Investigations Report Number 42, Version 4.0 (2004).[11] Specific laboratory analysis requirements include organic carbon measured by dry combustion, with adjustments for inorganic carbon based on acid treatment, and conversions to organic matter based on a common conversion factor (1.724) to calculate organic matter. The manual indicates that they also use loss on ignition (400 °C) to estimate organic matter. Bulk density measurements are also required, and numerous alternate measurement methods are listed.

3.4.3 Climate Action Reserve (CAR) – Soil Enrichment Protocol 1.0

The Climate Action Reserve, or CAR, protocol requires direct measurement of SOC by soil sampling and analysis via dry combustion to establish the baseline and for updates every five years. Bulk density (soil mass per unit volume) measurements are also required. It also requires that inorganic carbonates are accounted for in the laboratory analysis.

3.5 Potential Advances in Measurement, Reporting, and Verification

In general, significant advances in MRV technologies are needed, along with standardization. MRV is concerned with soil carbon stocks (mass carbon to fixed soil depth per area). Soil (organic) carbon stocks cannot be measured directly but are calculated from SOC concentrations and bulk density – both of these measurements are costly and subject to error. Therefore, advancements in improving soil carbon stock measurements, monitoring, and credibility are needed to reduce the costs associated with MRV. These advances need to address the following objectives:

- Fewer or better targeted soil samples through some combination of improved models, remote sensing, and in-field sensors;
- Increased speed, accuracy, and/or reduced cost of measuring SOC stocks and bulk density;
- Increased speed, accuracy, and/or reduced cost of confirming practices being implemented; and/or,
- Reduced cost of data management.

A variety of technologies and data sources are currently used to facilitate soil carbon management. There is a crucial need for continued advancement in the capacity of these tools to function in a more integrated, efficient way at the level of agricultural management (i.e., field, practice, animal feeding

[11] Note that this manual was superseded by Version 5.0 in 2014.

operation; Fig. 3.1). In the following, we outline some of the technologies and data sources that have the potential to improve cost effectiveness of MRV.

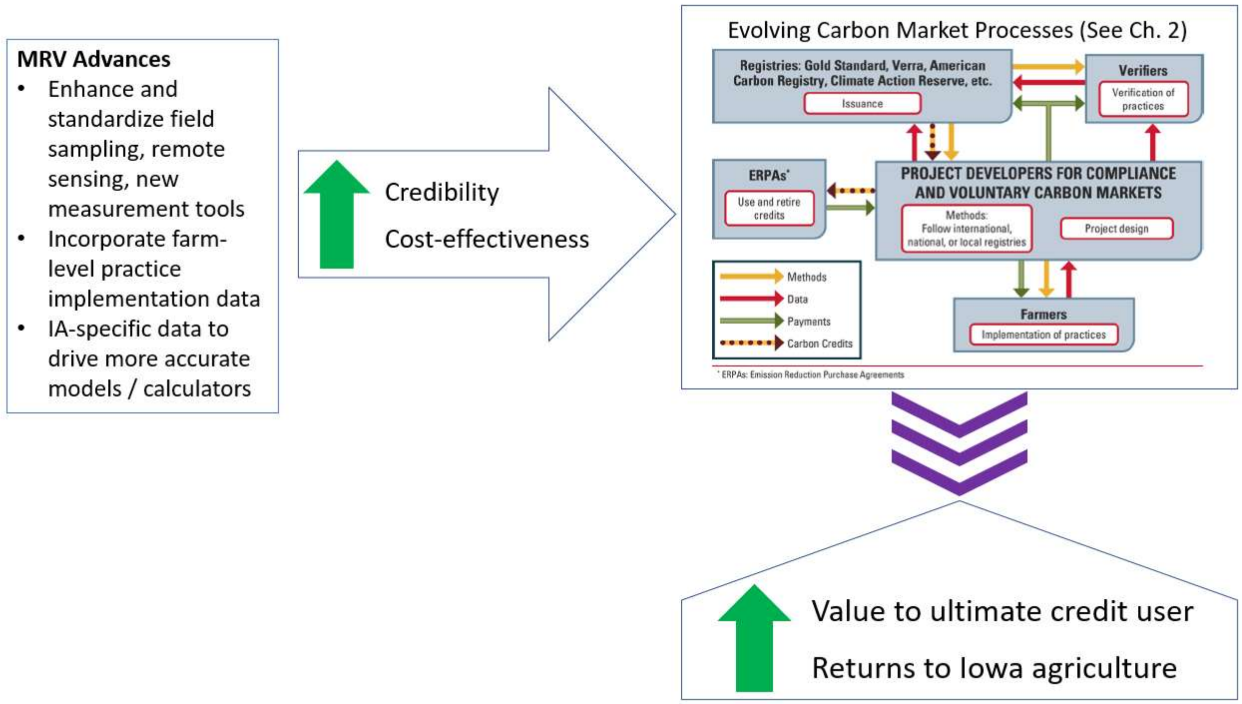

Figure 3.1. The seamless integration of multiple data sources and improved tools can support improved estimation, measurement, reporting, and verification and ultimately the market value of agricultural carbon credits.

3.5.1 Wireless Infrastructure

Advancements in the extent and type of rural internet service can help with data collection and monitoring costs to support both carbon market data needs and practice adoption. For example, in central Iowa, an at-scale platform for advanced wireless research is to be deployed across ISU's campus, the City of Ames, and surrounding research and producer farms as well as rural communities in central Iowa, spanning a rural area with a diameter of over 37 miles (60 km). This effort is led by ARA, which stands for Agriculture and Rural Communities and includes numerous local, regional, and national partners. It serves as a wireless living lab for smart and connected rural communities, enabling the research and development of rural-focused wireless technologies that provide affordable, high-capacity connectivity to rural communities and industries such as agriculture (ARA 2021). ARA is part of the National Science Foundation Platforms for Advanced Wireless Research (PAWR) program.

At its foundation is the deployment of advanced wireless platforms in real-world agriculture and rural settings, capturing the systems and environmental properties as well as the application and community contexts of rural broadband. For instance, ARA will feature the future of precision agriculture in both crop and livestock farms, involving automated ground vehicles as well as cameras and nanosensors. Contiguously covering hundreds of square miles of rural area, ARA serves as an at-scale, deeply-programmable infrastructure for rural wireless research in real-world settings. These new data platforms can provide novel inputs to improve the modelling and monitoring required for MRV.

3.5.2 Automation of Farm Management Practice Data Collection

Electronic verification data for agricultural practices and input utilization have been available for decades through on-machine monitoring of seed and fertilizer inputs, tillage practices, and yield outputs. Historically, these systems required clumsy management of individual data cards with non-standardized file structures. In the past half-decade, agriculture technology and connected vehicles have evolved significantly. In today's agriculture, machine and crop input data stream seamlessly through application programming interfaces (APIs) into secure and connected cloud environments. These interconnected clouds allow growers to compile electronic field reports, independent of their machinery or technology provider brand, and to summarize all aspects of agricultural practices.

These data systems have the capability today to provide agricultural practice verification that certifies specific nutrient management, crop management, and spatial production plans. Information stored in these cloud systems is easily portable through APIs or field reports to provide verification of specific agricultural practices or documentation of yield outcomes. One constraint of this system is that it relies on the presence of data to populate field reports. In the case of verification of no-till practices, no actual data are available to confirm this practice, and the lack of a data set should never be used as verification for a process.

With increasing use of context cameras on agricultural vehicles, from backup cameras on tractors to forward looking cameras for weed identification on sprayers, it is highly feasible to create automatic tillage and residue verification datasets in the future. Streaming of low-frequency ground photos to verify residue levels and tillage practices at various points in the season is increasingly feasible. In addition, increased use of high-resolution imagery can also be leveraged to significantly improve the capacity and throughput of sustainability verification processes.

Commercialization of these technologies is advancing quickly. For example, N-Sense (2021) is developing a soil sensor system based on Fourier transform infrared sensor technology that can be attached to a nitrogen fertilizer applicator and used to modulate side-dress nitrogen fertilizer application rates in real-time based on soil nitrate levels as the applicator moves through a field. This type of technology could eliminate the need for soil sampling for side-dressing and the time delay between sampling and fertilizer application, and also provide greater spatial resolution than is possible with soil sampling.

To successfully meet verification goals, carbon registries need to coalesce around a set of minimum required metrics to empower the agricultural technology industry to put in place systems that automate collection and create value in the supply chain. The agricultural technology industry is unlikely to invest in research and development to support carbon markets until the carbon industry aligns and clearly conveys critical measurement factors at the scale of management.

3.5.3 Field-Sampled Data

Standard methods for field-based monitoring of changes in soil carbon stocks and greenhouse gas emissions are onerous, relatively expensive, and prone to propagation of sampling and analytical errors. Technological advances are making measurement easier and less expensive, and in some cases may improve precision and accuracy. These new technologies allow for the expansion of measurements in time and space, and enable a higher rate of adoption by non-scientists (sometimes called community or participatory science).

3.5.3.1 Soil Carbon

As noted earlier, many carbon programs require some soil sampling and analysis. These programs generally specify that SOC is measured by dry combustion, which is time-consuming and expensive compared to the loss on ignition (LOI) method commonly used for measuring soil organic matter.

LOI data are sometimes used to estimate SOC through a conversion factor, traditionally 1.724 (e.g., a soil with 4% organic matter has approximately 2.3% SOC [4 / 1.724]).[12] The use of LOI data to accurately infer SOC concentration requires calibration specific to the variation of the LOI method being used, to the local Major Land Resource Area (MLRA) (Konen et al. 2002), and possibly calibration down to the specific soil series mapped at the location where the sample was collected. Additional Iowa calibration data for the LOI method are provided by Cambardella et al. (2001).

An important consideration is the fact that even though LOI data with calibration can correlate well with dry combustion data (Konen et al. 2002), it is less accepted by current carbon market protocols. It is worth noting that LOI data are still used extensively in scientific literature. Although there is a vast amount of existing LOI data, often held by farmers and ag retailers, sample collection and lab procedures vary, the level of calibration is often not known, and privacy concerns limit access to the data. A standardized and recommended LOI procedure is available (Nelson and Sommers 1996), but scientific debates continue on the temperature and duration of the test that will ensure reliable results. If these shortcomings were addressed, it is plausible that future LOI data could be beneficially used to inform programs and models, support general carbon project planning aid, and/or be coupled with dry combustion data to reduce analytical costs.

To maximize the utility of either LOI or dry combustion data in developing better carbon models or calculators, sample data need to be georeferenced—ideally from a single sampling location rather than a composite sample. In contrast, the typical practice for both routine soil fertility sampling and some registry protocols is composite sampling. Typical ag retail soil fertility sampling that includes LOI data is relatively shallow compared to registry protocols. Moreover, as noted in **Chapter 4**, data are likely needed at even greater depths than required in some protocols to fully account for changes in soil carbon stocks.

Bulk density is another soil parameter of great importance to quantifying SOC stocks. Bulk density is simply the mass of soil per unit volume. SOC is a concentration, or mass per unit mass. Both factors need to be measured (or assumed), as well as the depth of sampling, to establish the total stock of carbon on any plot of land. As noted previously, some of the registry protocols require bulk density. The uncertainties associated with bulk density measurements are even greater than those associated with SOC measurements and can markedly increase the total error in measurement. Bulk density can be measured in the field, but it is difficult and time-consuming to obtain accurate measurements. Bulk density is not well suited for the collection of large quantities of accurate data quickly with present technology. One potential approach is the use of "pedotransfer functions," which means the use of other soil parameters to estimate other parameters. Software programs, such as SPAW (2017), can be used to approximate bulk density from organic matter or SOC and soil texture. These estimates may not be fully accurate, but could provide a means of consistency across programs. Challenges with the

[12] Research indicates the traditional "rule of thumb" to be lacking, and that this factor is too low for most soils. A general factor of two is likely more accurate for most cases (Pribyl 2010).

pedotransfer function approach include possible errors in measuring the factors used to estimate organic matter or SOC, coupled with the fact that measuring the predictor variables required (e.g., soil texture) is typically not part of registry protocols.

It is also important to note that simple considerations, such as variations in the depth of sampling, could exceed variability in analytical methods. To demonstrate a measurable increase in SOC, it will be necessary to document an increase that is larger than the “error” associated with the calculation. In other words, if carbon credits are based on measurements of soil carbon stocks, both analytical and sampling uncertainties will determine whether those credits are credible. The larger the initial stock of carbon, the more carbon must be stored to get beyond the uncertainty limits. This will be an issue in Iowa soils, where carbon stocks are already high.

One area of technological advancement with respect to monitoring SOC is tools that can quickly measure soil properties that indicate the amount of SOC present without requiring sampling and laboratory analysis. One of the most common emerging tools is spectroscopy—both in the field and in the laboratory—to estimate SOC (Gregory et al. 2006; Gomez et al. 2008; Minasny et al. 2011; Seybold et al. 2019; Shi et al. 2015). Soil organic matter and soil minerals absorb light at different wavelengths, enabling estimation of a number of properties (Dangal et al. 2019; Wijewardane et al. 2020). A few studies have confirmed the accuracy of these approaches (Paul et al. 2019; Sanderman et al. 2020). One research team even used mobile phone cameras to estimate SOC (Fu et al. 2020). Dual technological developments in spectroscopy and ubiquity of high-resolution cameras on mobile phones make this technology one of the most promising. A few protocols (e.g., FAO (2020), not used in Iowa to our knowledge) will accept SOC data from multi-spectral probes (Oldfield et al. 2021), suggesting the potential that other registry protocols could accept these data in the future.

Some of these spectroscopic tools are already being commercialized by companies such as Yard Stick and ChrysaLabs. The Yard Stick (2021) approach measures infrared light reflected from soil and also includes a resistance sensor for approximating bulk density (Oldfield et al. 2021). It was based on a research and development grant from the United States Department of Energy’s ARPA-E Smartfarm program, and was reportedly based on work by Wijewardane et al. (2020). The ChrysaLabs (2021) company is marketing another spectroscopy-based probe, claiming to be able to measure nitrogen, potassium, phosphorus, pH, soil moisture, cation exchange capacity, and minor nutrients in addition to soil organic matter. The applicability of these tools to carbon markets in Iowa is yet to be established, but pilot testing is currently underway by multiple entities directly engaged in carbon market programs.

In yet another approach still under development, Kavetskiy et al. (2019) demonstrated a neutron-gamma analysis method for mapping soil carbon, where a golf cart tows a trailer with a scanning system across fields to collect detailed data in transects without soil disturbance.

Other technologies on the horizon, both in nascent stages or undeveloped as of yet, can change how we monitor changes in SOC. Verifying that management practices do indeed increase SOC under a variety of environmental conditions will require further improvement in precision and ease of measurement.

3.4.3.2 Methane

There is a high level of current interest, both in the United States and internationally, in limiting methane emissions as the low-hanging fruit for reducing GHG. This interest is driven by a number of factors, including its high global warming potential (GWP; 25 times that of carbon dioxide; see **Appendix**

11.2 for an explanation of GWPs), comparatively short lifespan in the atmosphere (~10 years), and the existence of numerous point sources that could potentially be controlled and/or measured. This interest in controlling methane, particularly by compliance markets, such as the Low Carbon Fuel Standard (LCFS) in California,[13] creates an opportunity for Iowa agriculture, because the reduction of methane emissions from livestock operations can have considerable value in the market, e.g., through the production of renewable natural gas (RNG) from manure or plant-based biomass. As noted in **Chapter 5**, farms with liquid manure handling systems converting to anaerobic digestion are best positioned to engage in these markets. This engagement would necessarily entail containment of methane gas emissions and ongoing monitoring of containment, treatment, and gas conveyance systems. An overview of research tools, baseline establishment, and technologies for ongoing project monitoring will be discussed in this section.

Enteric emissions from livestock can be measured, but typically only in a research context, such as through the use of eddy flux towers and open path lasers measuring emissions from animal groups, or direct measurement of the breath of cows (Todd et al. 2019). (Additional details on research tools to measure enteric emissions are provided in **Chapter 5**). Methane emissions from manure pits can be measured using open path lasers together with data on the ventilation system capacity. Self-contained laser units can also be used to measure methane, carbon dioxide, hydrogen sulfide, and ammonia.

The data in IPCC (2006, 2019) or models validated for accuracy can be used to establish the baseline enteric methane emissions of livestock operations (models are described further in **Chapter 5**). Baseline values for methane emissions from manure and manure storage systems lagoons typically come from data the IPCC (2006, 2019) has compiled, sometimes coupled with ASAE (2005) data. The IPCC reference values are based on data collected in the 1980s (Steven Trabue, USDA-ARS, *personal communication*) and/or models, such as GREET for the California LCFS market. Surface area and manure age are key factors for projecting methane emissions. Manure sampling to project emissions is based primarily on measuring volatile solids in the manure. As noted in **Chapter 5**, there is considerable facility-to-facility variation in the emissions from animal operations, and more data, as well as more cost-effective ways to collect these data, are needed to establish accurate baseline values for specific farms. (Additional details on estimates based on reference values for manure emissions are provided in **Chapter 5**.)

Much like the conventional natural gas industry, leaks represent lost revenue for renewable natural gas (RNG) applications, which may be a stronger driver to control leaks for some operators than offset program compliance needs. Methane leak detection is an ongoing, often labor-intensive task for large operations. Windy conditions disperse methane, limiting the ability to detect small leaks. Underground leaks often disperse before they can be detected. The California LCFS program assumes it is impossible to reduce leaks from animal waste RNG systems to below 2%.

A number of technologies are available for facility leak detection, including optical leak imaging (infrared cameras), laser leak detector systems, simple soap bubble screening, organic vapor analyzers/toxic vapor analyzers, and acoustic leak detection (CCAC 2015). Optical leak imaging is a common method for detecting small leaks and would likely be the approach that manure/biomass system owners would use to detect and manage leaks in contained systems, such as covered lagoons, covered pits, and anaerobic

[13] There is a separate California Cap and Trade System for Power Plants, which also allows carbon offsets from methane capture and destruction projects (discussed in **Chapter 2**).

digesters, and that third-party verifiers would use if onsite. There is considerable interest in continued development of better detectors.

There are specific protocols for several different methane-reducing practices. In addition to protocols established by the LCFS (e.g., CARB 2014), the Gold Standard Registry, for example, provides a protocol for the use of feed supplements in dairy cows (Gold Standard 2019).

3.5.3.3 Nitrous Oxide

Carbon programs vary in exactly how they deal with nitrous oxide. Some do not include it, but essentially all are at least in the process of evaluating how it can be included. There is no simple way to monitor or measure nitrous oxide at the farm or field scale. There are a variety of research-based tools that are necessary to gather the detailed data needed to inform models. These research methods are not feasible or cost-effective for carbon project developers or their third-party verifiers. These research tools include sampling chambers placed over the soil surface (Lawrence et al. 2021; McDaniel et al. 2019) and eddy flux towers (Nemitz et al 2018). Soil flux chambers are necessary to evaluate the influence of specific soil properties or treatments at the plot scale (i.e., a few square yards or tens of yards). Eddy flux towers collect data on gas fluxes and resemble a small weather station. They integrate data over a potentially much larger footprint than sampling chambers (e.g., multiple acres) and provide continuous monitoring data (compared with discrete, manual sampling via flux chambers). In addition to nitrous oxide (Liang et al. 2018), this technology can be used to measure carbon dioxide and methane alongside landscape water and energy fluxes. Nitrous oxide emissions from manure, both for baseline and ongoing emissions, are typically based on the IPCC Tier 2 approach (IPCC 2006, 2019), sometimes coupled with data from ASAE (2005). (Additional details on nitrous oxide emissions from manure are provided in **Chapter 5**.)

3.5.4 Remote Sensing Data

3.5.4.1 Remote Sensing of Management Practices

The adequacy of remote sensing to track soil processes and management practices relevant to carbon markets—such as crop type, residue cover, and cover crops is not yet satisfactory, although advancing. Crop type mapping has been operationalized by the USDA through the Cropland Data Layer (CDL) product, offered by the National Ag Statistics Service (Boryan et al. 2011), providing accuracies in the 85–95% range primarily for large-acre crops. The Cropland Data Layer is trained on the large amount of survey data collected by the USDA as part of their crop monitoring mission. For Iowa, the published 2020 data on primary crops, such as corn and soybeans, have an accuracy of 93%. Forage crop accuracy was much lower, at 26%, caused primarily by confusion with alfalfa, other hay crops, and pasture/ grasslands.

Remote sensing of residue cover and cover crops is not operationalized beyond the Operational Tillage Information System (OpTIS) and is much less advanced. OpTIS provides low resolution, large watershed-scale estimates of tillage classes. Residue cover is frequently used as a proxy to estimate tillage practices under the assumption that reduced residue cover is indicative of more intensive tillage, although this is not always the case, e.g., consider the practices of stover harvest and moldboard plow which both result in very low residue levels. These confusions may eventually be discriminated by radar remote sensing. The current state-of-the-art in residue cover mapping involves calibrating satellite observations from Landsat and Sentinel 2 satellites with ground measurements (Laamrani et al. 2020) or estimates from other satellites with increased spectral resolution in the shortwave infrared spectrum (Hively et al. 2019). Similarly, remote sensing of cover crops relies on calibration with ground measurements of confirmed cover crops (Rundquist and Carlson 2017; Laamrani et al. 2020). However, there is a possibility for using guided big data approaches to determine the low and high residue cover thresholds (Beeson et al. 2020) or cover crops by using Normalized Difference Vegetation Index (NDVI) thresholds. Figures 3.2 and 3.3 illustrate the use of remote sensing data to estimate crop residue cover at state-wide and field scales, respectively.

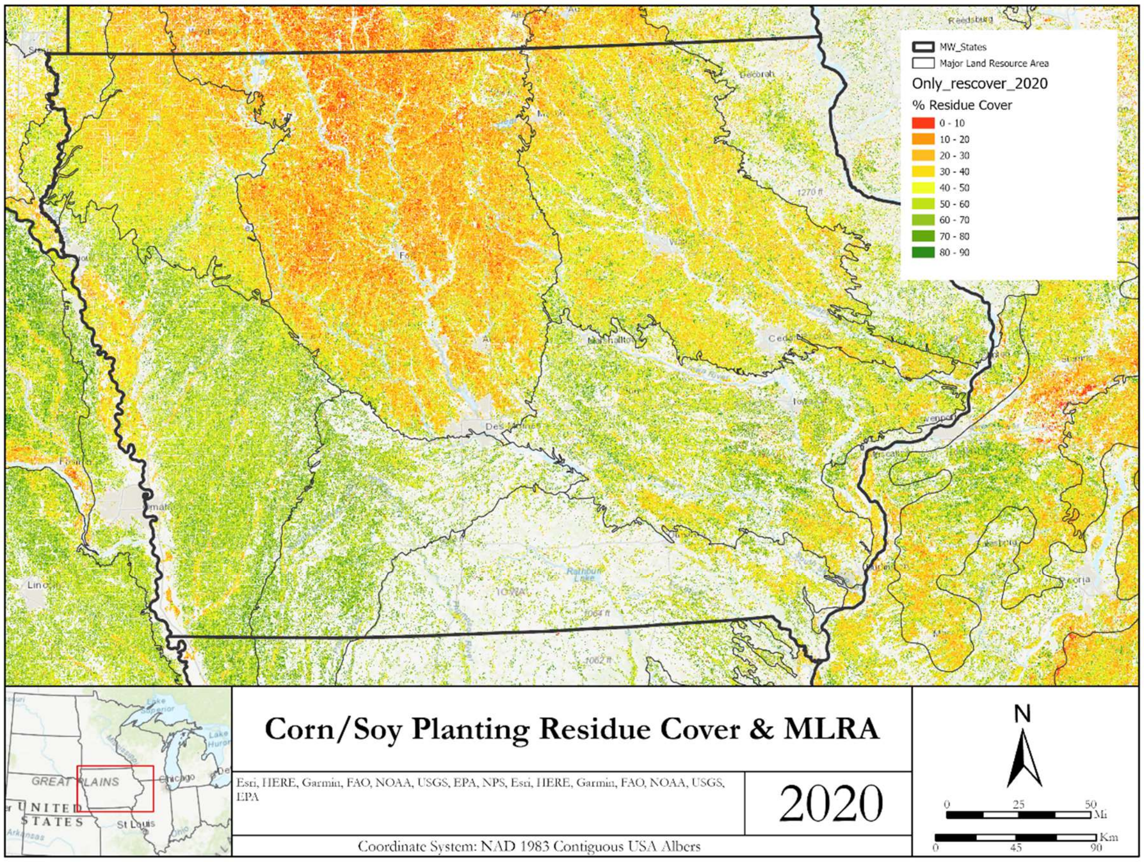

Figure 3.2. Estimated crop residue cover by landform region for Iowa based on remote sensing tools (spring 2021).

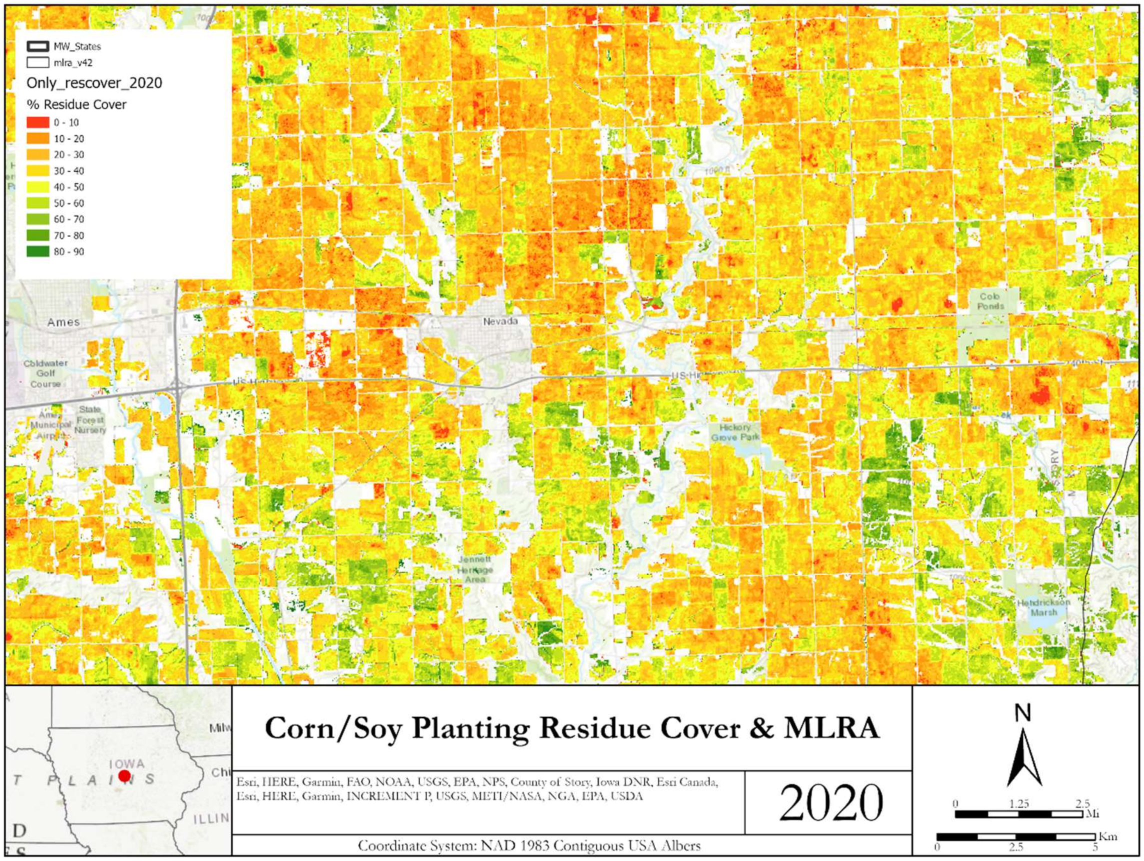

Figure 3.3. Example estimated field-scale crop residue based on remote sensing tools (spring 2021).

Research indicates that soil erosion has caused loss of SOC stocks, yet the quantification of SOC redistribution and loss across the landscape is lacking, especially for smaller parcels such as buffers and waterways. Additionally, if remote sensing will be used in whole or in part to verify carbon storage practices, there is a need for collecting additional datasets on residue cover and cover crop validation from Iowa as the current datasets comprise only a few years across less than 20% of the state. These data should preferably be shared openly to facilitate use by the remote sensing community for evolving the detection of residue cover and cover crops. Research is ongoing into the use of machine learning models to streamline data collection, analysis and verification processes for cover crops, cash crop planting date, and cash crop tillage practices. Currently the data volumes are growing and the lack of knowledge of how to process it into usable information as well as the time required are bottlenecks. With the advent of satellite constellations, such as Planet's Dove constellation, near daily imagery at 9.8-ft (3-m) resolution in multiple bands is now available. These data can revolutionize the process of detecting cover crop planting, cover crop biomass, and cash crop planting and tillage dates.

3.5.4.2 Remote Sensing of Changes in Carbon Stocks

Currently, it is impossible to measure changes in soil carbon stocks using remote sensing methods. Over a decadal timescale, it is possible to measure changes SOC associated with changes in soil location, caused primarily by water erosion but also by wind erosion using LiDAR.

3.5.4.3 Remote Sensing of Greenhouse Gas Emissions

There are multiple programs underway to obtain more detailed data using remote sensing of specific sources of carbon dioxide and methane. Remote sensing can currently not be applied for nitrous oxide sensing. Some of these programs claim that the level of detection and spatial resolution will be at the "facility level," so that it will be possible to identify specific point sources of some greenhouse gas emissions. These data may be used for a variety of purposes.

One such program is Carbon Mapper, a non-profit organization whose collaborators include the NASA Jet Propulsion Laboratory, Planet Labs, University of Arizona, CARB, Rocky Mountain Institute, and several philanthropies (Carbon Mapper 2021). Two demonstration satellites with carbon dioxide and methane monitoring capabilities and global coverage organized by this group will be coming online in 2023, and there will be a "full constellation" of these satellites by 2025 (Schingler 2021). Oil and gas, livestock, and landfills are initial areas of focus. The level of detection is reportedly an emission rate of 0.1 Mt/h, with a spatial resolution of 30 m (98 ft) (NAS 2021). The emission rate of 0.1 Mt/h equates to 876,000 kg/year (2 million lb methane per year) over an area 30 m × 30 m (98 ft × 98 ft). A typical swine facility in Iowa (5000 head) will produce approximately 61,000 kg/year (134,000 lb) of methane, far below the detection limit of this technology. The methane emissions of a very large dairy facility (3000 head or larger, producing around 300 kg methane per cow or 900,000 kg methane per year) could potentially be detectable, but the majority of dairy operations are smaller than this threshold. Therefore, this technology at present has very limited if any impact on MRV for Iowa livestock operations. In comparison, current satellite technology is limited to detecting an emission rate of 10 Mt methane per hour at a spatial resolution of 7 km (4.3 miles) (NAS 2021).

NASA will also be launching the Geostationary Carbon Observatory (GeoCarb), targeted for launch in 2022 (eoPortal 2021), that will fly in a geostationary orbit. Its longitude will allow "wall-to-wall" observations over the Americas between 50 degrees north and south latitude—from the southern tip of Hudson Bay to the southern tip of South America. GeoCarb will collect 10 million daily observations of carbon dioxide, methane, and carbon monoxide concentrations and solar-induced fluorescence (SIF) at a spatial resolution of approximately 3 to 6 mi (5 to 10 km) (NASA 2021).

3.5.5 Modeled Data

Carbon project calculators, such as COMET-Farm, are used both as decision support tools during initial planning and to support MRV (see **Chapter 8** for calculator descriptions). A project implementer (e.g., a farmer) may use these calculators to determine whether to enter into a carbon program or not. Baseline, past, current, and future practices to be implemented are typically analyzed and documented using tools, such as COMET-Farm, to establish the projected change in CO_2e as the basis for program enrollment. Calculators are also used to fill in spatial and temporal gaps in field sampling to support monitoring or verification. For this reason, robust agroecosystem calculators are needed to support the growth of the carbon market. At present, these calculators are better at predicting potential maximums and averages, especially over broad spatial extents, than true values (**Chapter 8**). While calculator predictions may be accurate on average, accurately estimating carbon credits at the scale of management (i.e., field, practice, animal feeding operation) for individual farm operations is challenging. Thus far, the accuracy of calculators for use in monitoring and verification has not been an impediment to market development. Example output from a COMET-Farm analysis is provided in **Chapter 8**.

3.5.6 Other Data

Other existing and emerging sources of public data could be used to support MRV directly or indirectly through improved models. Some of these include Conservation Reserve Program data, Iowa Nutrient Reduction Strategy and related data, and the Agricultural Conservation Planning Framework (ACPF) and Iowa State University Best Management Practices (ISU-BMP) database.

3.5.6.1 Conservation Reserve Program Data

Over the next few years, extensive additional data on perennial systems will become available and inform models and programs. The United States Department of Agriculture (USDA) is investing $10 million in a new initiative to sample, measure, and monitor soil carbon on Conservation Reserve Program (CRP) acres to better quantify the climate outcomes of the program. This initiative was launched in October 2021. The CRP Climate Change Mitigation Assessment Initiative projects will survey, sample, and measure the climate benefits of land enrolled in CRP conservation practice types over time to help USDA better target CRP practices and strengthen conservation planning tools. These models include the Daily Century Model, or DayCent (**Chapter 9**), which simulates the movement of carbon and nitrogen through agricultural systems and informs the National Greenhouse Gas Inventory. Data will also be used to strengthen the COMET-Farm and COMET-Planner tools, which enable producers to evaluate potential soil carbon storage and greenhouse gas emission reductions based on specific management scenarios.

USDA partners will conduct soil carbon sampling in three categories of CRP practice types: perennial grass, trees, and wetlands.

- **Perennial grasses**: In consultation with USDA, Michigan State University will sample and measure soil carbon and bulk density of CRP grasslands.
- **Trees**: Mississippi State University will partner with Alabama A&M University to collect above- and below-ground data at 162 sites across seven states documenting CRP-related benefits to soil and atmospheric carbon levels. Information will help further calibrate the DayCent model. This five-year project will focus on the Mississippi Delta and Southeast states.
- **Wetlands**: Ducks Unlimited and its partners will collect data on carbon stocks in wetland soils as well as vegetation carbon levels at 250 wetland sites across a 15-state area in the central US. Data will support DayCent and additional modeling.

3.5.6.2 Iowa Nutrient Reduction Strategy and Related Data

Estimates of the current extent of conservation practices relevant to carbon markets are provided in **Chapter 7**. In brief, the current statewide efforts to track conservation practices (e.g., cover crops, tillage practices, fertilizer management, land use, and structural practices) are based primarily on water quality concerns and are driven by the Iowa Nutrient Reduction Strategy. An extensive private sector effort, led by the Iowa Nutrient Research and Education Council (INREC) and data analysis provided by ISU, collects data on farm management practices through agricultural retail partners.

3.5.6.3 Agricultural Conservation Planning Framework and Iowa State University Best Management Practices Databases

The ACPF and ISU-BMP database are the largest known open databases on field-level land management practices and can provide information on cropping practices for the last 10 years and the locations of BMPs across the state as of 2010. Figure 3.4 illustrates the type of data available in the BMP database. The ACPF has become a key tool for watershed and conservation planning across Iowa to address water quality needs (see also **Chapter 8** for additional information on ACPF).

The ACPF incorporates detailed annual land use data from the National Agricultural Statistics Service (NASS) CDL going back to 2010 for the entire state of Iowa which can be used to inform carbon credit programs and state and regional planning efforts around CO_2e issues and opportunities. When combined with remote sensing of residue cover done for the Daily Erosion Project, the ACPF can also provide tillage class estimates (Gelder et al. 2018).

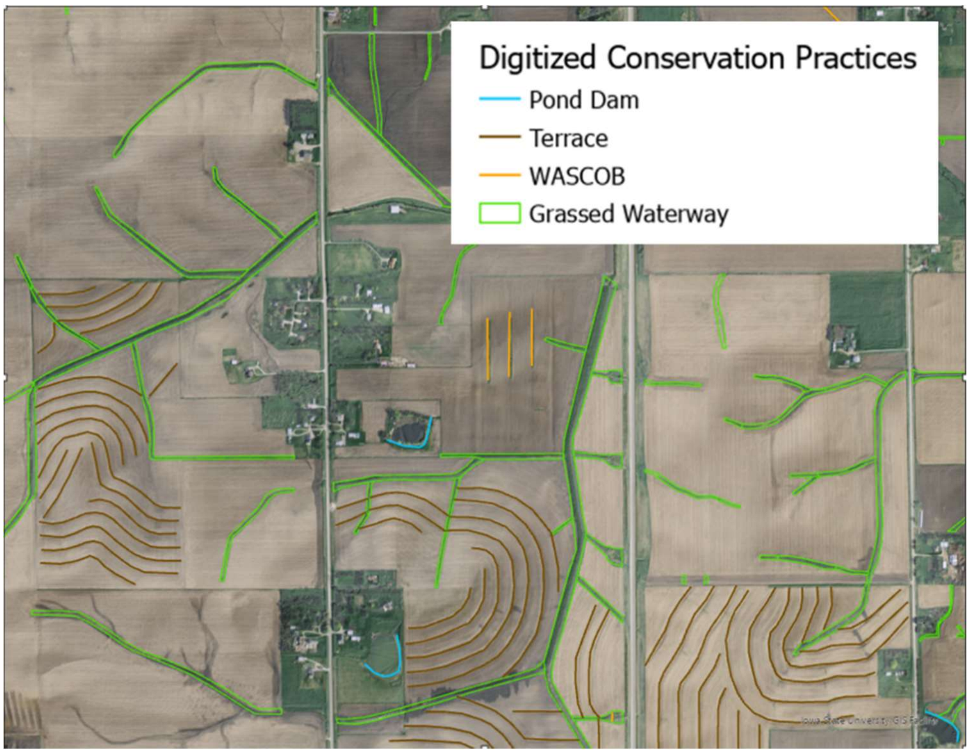

Figure 3.4. Example BMP database information for existing structural conservation practices for Blackhawk County, Iowa.

3.6 Putting it All Together

Technological advances in automation of management data collection, rapid field-based measurements, remote sensing, modeling, and other tools offer the promise of increasing the amount of information available and of lowering the cost for carbon market participants. To reach the goal of more credible, and therefore more valuable carbon credits, these technologies need to be put together in a seamless way. A number of areas require attention to achieve this goal:

Technology needs:

- Advances in remote sensing and data analysis to provide more accurate data on tillage and cover cropping (e.g., planting date and level of growth impacts outcomes). A major need is additional ground truthing to better calibrate these tools.
- Iowa data are needed on multi-spectral probes and other technological innovations for rapid measurements of SOC.
- Carbon market programs need guidance on sampling Iowa soils for SOC. This could be informative (e.g., understanding tradeoffs between sample number, depth, location, analytical method, importance of bulk density, etc.) without being prescriptive (i.e., allowing market requirements to continue to evolve).
- For predicting nitrous oxide emissions based on different practices, it would be helpful to develop a semi-empirical regression model based on parameters such as fertilizer (rate, placement, timing and form), SOC, soil texture and other soil properties, and climate.
- The flow of data from farm machinery-based sensors could be streamlined to meet the needs of carbon market protocols with the assistance of farm equipment manufacturers and ag retail entities.
- Standardize methods and calibration for the widely used and low-cost LOI organic matter data farmers and ag retailers have long collected to use at least as a planning tool, and possibly to support lowering the cost of carbon program monitoring and improving supporting models and calculators.
- Recognize the challenges inherent in measuring soil bulk density accurately and quickly, and consider adopting alternate approaches, such as pedotransfer function models, to provide standardized estimates of SOC.
- Farmer innovation is a powerful tool to support the development of new technologies and approaches, but currently inadequately incorporated into current programs, models, and calculators.

Data needs:

- Well targeted sampling and analysis on a wider range of soil types and landscape to allow development of better models. Data are particularly limited at present in the following areas: field-scale nitrous oxide emissions, SOC dynamics below the surface 12 in (30 cm) (**Chapter 4**), and impacts of drainage and climate changes on SOC and greenhouse gas emissions.
- Reliable soil maps are critically important tools for both landowners and policymakers. The number of soil samples collected in support of the Cooperative Soil Survey Program has been declining over the past 40 years because of the general perceptions that soil does not change and that all areas of Iowa have already been mapped. Ongoing work by Dr. Bradley Miller is improving digital soil mapping techniques to make the process more efficient, particularly in reducing the density of samples needed to produce reliable maps. The goals of this work are to enable the stratification of historical observations into finer time steps and to produce maps for more soil properties than the soil series of the current maps. Strategic sampling of the current soil landscape coupled with remotely sensed data of many kinds is needed to provide better understanding of the processes creating spatial patterns of soil properties and to update all soil

maps. Research-based improvements in digital soil mapping will facilitate accounting for SOC stocks across entire hillslopes instead of individual points on the landscape.

Data management needs:

- A number of general data management issues need attention, but are not addressed in this report. These include questions of reducing data management costs, portability, ownership, security, privacy, and other data factors that impact the efficiency and credibility of the market, support model and calculator accuracy, and influence farmer participation.
- Carbon registries need to coalesce around a set of minimum required metrics to empower the agricultural technology industry to put in place systems that automate collection and create value in the supply chain. The agricultural technology industry is unlikely to invest in research and development to support carbon markets until the carbon industry aligns and clearly conveys critical measurement factors at the scale of management. Additional research may be required to establish these metrics.

Partnership needs: There is a clear current and ongoing need for close collaboration of agronomists/soil scientists, statisticians, data management experts, farmers, agriculture retail, carbon market brokers, and others to design and help standardize efficient and credible ways to gather required data. Examples of these needed collaborations include:

- Detailed data collected on nitrogen application rates by agriculture retailers through their data management systems. A path forward is needed to address privacy and business/proprietary considerations such that these data can be accessed to improve both SOC and nitrous oxide models. As noted in **Chapter 4**, soil carbon and nitrogen processing are intimately linked.
- Iowa State University researchers could partner with carbon calculator developers, such as Colorado State University, to help incorporate Iowa-based research into these tools and models.
- There is a need for demonstration of a fully transparent system for generating, measuring, reporting, and verifying carbon credits. A public-sector farm, such as an Iowa State University Research and Demonstration Farm, could be a credible location for such a demonstration.
- The pathway for implementing faster/better/cheaper MRV based on scientific advances goes through the carbon registry protocols. Iowa institutions need to be part of a very broad-based effort to bring advances in MRV forward through an international-scale, public-private partnership effort, engaging government, university, private sector, nongovernmental organizations, and philanthropy.
- Soil moisture and temperature data are needed to inform SOC and related GHG models. Networks that collect soil moisture data, such as rain and soil moisture gauges in the University of Iowa Flood Information System and the Iowa State University Soil Moisture Network, could likely be better linked to these models.

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Chapter 4. Cropping Systems and Land Management

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4.1 Highlights

- Crop production and soil management can both increase and decrease overall agricultural greenhouse gas (GHG) emissions.
- Improved management of nitrous oxide emissions from cropland will have greater impacts on CO_2e emissions than land management focused solely on storing carbon in soils.
- Farmers can increase the amount of carbon stored in many Iowa soils by reducing tillage, adding more crop residues, using winter cover crops, extending crop rotations, or converting to perennial crops or conservation practices.
- Careful management of the rate, timing, and placement of nitrogen fertilizer application can minimize the release of nitrous oxide from most agricultural soils.
- Installation of tile drains in poorly drained soils can reduce nitrous oxide emissions but may also increase carbon dioxide emissions, thus resulting in net soil carbon loss.
- Edge-of-field practices installed to improve water quality (e.g. saturated buffers, bioreactors, wetlands) are not likely to emit more nitrous oxide and methane than their associated croplands.

4.2 Background

Photosynthesis by green plants can remove carbon dioxide from the atmosphere at a scale that is responsive to the challenges posed by climate change (NASEM 2019; Richard 2021). Iowa—by virtue of our favorable climate, fertile soils, and proven capacity to innovate—is well-positioned to take advantage of emerging carbon markets through crop and land management. The Iowa climate generally provides sufficient moisture to allow intensive row cropping without irrigation, along with intense solar radiation during the growing season. Iowa has some of the highest gross primary production (a measure of photosynthetic rate, hence rate of carbon dioxide removal from the atmosphere) in the world (Gaunter et al. 2014; Fig. 1.3). Iowa further has some of the world's most fertile soils, due to a combination of relatively young soils, favorable climate, and the native prairie that formerly covered much of the state. Iowa is also a proven center of agricultural innovation, driven by a powerful combination of farmer creativity, public and private research capacity, and industry support.

Agronomists evaluate how crops perform by considering the interaction of genetics, environment, and management. Iowa has a long history of advances in crop breeding (genetics), which could be brought to bear on optimizing crop production with the goals of carbon markets. As noted above, Iowa has a highly favorable environment, specifically the climate and soils necessary to optimize photosynthesis. Iowa has a proven history of using technology and management to achieve increasingly large crop yields while leveraging increasingly scarce labor resources. Iowa also has a proven ability to innovate with conservation practices, leading the way nationally in developing new practices such as riparian buffers, prairie strips, bioreactors, and saturated buffers.

Iowa's agronomic prowess intersects with advances in agricultural equipment. Advances in artificial intelligence and machine vision are rapidly increasing the level of autonomy present in current and future machines. Machines have the potential to sense nearly every aspect of agronomic management. Implementing state-of-the-art agronomic practices to align with carbon markets effectively and profitably will require these new sensing and control capabilities. For example, improvements in on-machine sensing, coupled with large supporting databases, will help to directly predict soil fertility needs and potentially soil carbon levels. These innovations, along with improved seasonal weather forecasts, can enable site- and season-specific decisions on fertilizer application and seed management, opportunities for direct reduction in total fertilizer use, and on-the-go optimization of economic and sustainability metrics in crop production.

In this chapter, we summarize current understanding of practices associated with cropping systems and land management that can be applied in Iowa to reduce GHG emissions associated with agriculture and to remove carbon dioxide from the atmosphere. Our goal is to identify agricultural practices that will promote high-quality, credible agricultural carbon credits. The agricultural management practices considered have either been tested, used in the past, or are presently used in Iowa. The capacity factors mentioned above, combined with new publicly funded scientific research on carbon and GHG in agroecosystems, can be brought together to optimize how farmers manage Iowa's cropland for multiple objectives, including for carbon markets.

4.2.1 Reduction of CO_2e in the Atmosphere

Reducing GHG emissions associated with the agricultural sector slows down the rate at which CO_2e are released to the atmosphere. In 2019, the agricultural sector of the United States contributed 10% of total national GHG emissions, driven largely by carbon dioxide (CO_2) from soil carbon loss and fossil fuel use, nitrous oxide (N_2O) from nitrogenous fertilizer use, and methane (CH_4) from ruminant livestock production (USEPA 2021). Overall, agricultural GHG emissions in the United States have increased by 12% since 1990 (USEPA 2021). Nitrous oxide emissions associated with fertilizer use account for more than 50% of GHG emissions by the United States agriculture sector and contribute more than 5% of total nationwide emissions. Nitrous oxide is the primary contributor to agricultural GHG emissions because nitrous oxide is a powerful GHG: one ton of nitrous oxide traps 298 times more heat in the atmosphere than one ton of carbon dioxide over a 100-year timescale (Myhre et al. 2013; USEPA 2021; see **Appendix 11.2** for more information on the GWPs of GHGs). Moreover, for every pound of fertilizer applied, approximately 4 lb of CO_2e are released to the atmosphere as a result of manufacture, transport, and application of nitrogen fertilizers (independent of nitrous oxide emissions after application) (Schlesinger 2000).

It is crucial to understand the relationships between carbon and nitrogen dynamics in soils and the impacts of agricultural management practices on these dynamics. Climate-focused agricultural policy in the past has tended to emphasize practices focused on soil carbon. Yet ongoing nitrous oxide emissions from Iowa agricultural soils have a larger climate impact than emissions of carbon dioxide from soil or plant residues. Soil drainage class plays an important role. Central Iowa soils that are not well drained have been shown to release more nitrous oxide than well drained soils in other parts of the Midwest where previous research has been conducted (Lawrence et al. 2021). Moreover, long-term measurements from typical corn/soybean rotations on poorly and somewhat poorly drained soils in Central Iowa indicate that the typical CO_2e of nitrous oxide emissions are approximately two-fold greater than CO_2e removals that might be achieved over comparable periods through adoption of no-till and cover crops (Parkin and Kaspar 2006; Parkin et al. 2016; Ogle et al. 2019; McClelland et al. 2021). These findings have been recently confirmed by comprehensive field measurements and a literature synthesis conducted by Iowa State University researchers (Lawrence et al. 2021).

Acknowledging the typically high nitrous oxide emissions characteristic of Iowa corn/soybean production systems has critical implications for the development of carbon markets in Iowa. First, we recognize that adoption of best practices for increasing soil carbon storage will by itself be insufficient to counteract the ongoing climate change effects of nitrous oxide emissions from corn-soybean production systems. Second, practices aimed at reducing nitrous oxide emissions may have greater positive impacts on CO_2e budgets in some agricultural production systems than will carbon management practices. This is especially important given that increases in soil carbon are potentially vulnerable to loss following potential future changes in management or climate.

Reducing nitrous oxide emissions from Iowa farms and capitalizing on related opportunities in carbon markets will require focusing on increasing the efficiency of nitrogen fertilizer use at field and sub-field scales. This includes the "Four R's" of nitrogen fertilizer management—the Right timing, Right placement, Right source (or fertilizer type), and Right rate. In addition, the rigorous evaluation of commercially available products that may suppress nitrous oxide emissions, such as biochar and enhanced-efficiency fertilizers (Zhang et al. 2020; Maaz et al. 2021). Optimizing nitrogen fertilizer timing (when plant nitrogen uptake is greatest) and placement (closer to the crop root) can mitigate some nitrous oxide emissions. Using coated or organic nitrogen fertilizers rather than sources that are more susceptible to loss can add additional mitigation. Fertilizer recommendations for using the 'right' rate, however, remains a scientific challenge as this nitrogen fertilizer rate changes from year-to-year and even within a field. Using a combination of soil and plant testing, remote sensing, and modeling is the current best option-toward predicting crop-nitrogen needs. Further development of precision farming equipment and spatially resolved fertilizer rate recommendations may be one way to more precisely manage agricultural inputs, such as nitrogen fertilizers, while maintaining farm profitability.

Nitrous oxide is produced in soil and water and released to the atmosphere as a byproduct of microbial metabolism. When nitrogen availability exceeds plant demand, it becomes increasingly likely to be lost as nitrous oxide, and nitrous oxide emissions tend to increase strongly (and non-linearly) as nitrogen fertilizer inputs increase (Shcherbak et al. 2014). Reciprocally, a 10% reduction in fertilizer nitrogen inputs is likely to lead to a greater than 10% reduction in nitrous oxide emissions, with additional co-benefits for water quality and farm profitability. Although carbon markets to date have largely focused on soil carbon as opposed to nitrous oxide, accumulating scientific evidence points to the critical role of addressing nitrous oxide emissions to achieve climate sustainability of Iowa agriculture. By incorporating fertilizer nitrogen management in the calculations of CO_2e in its carbon markets, Iowa can be a national leader in climate-smart agriculture.

4.2.2 Carbon Removal from the Atmosphere and Storage in Soil

To ensure the most effective removal of carbon dioxide from the atmosphere, agricultural management practices should promote persistent storage in soil. In other words, some fraction of organic matter inputs to the soil must be less susceptible to biochemical reactions that would release carbon dioxide or methane to the atmosphere within decades. **Persistence** refers to the ability of some soil organic matter to avoid decomposition by microbial enzymes. Several factors can help increase the net stock of stored soil carbon, including characteristics of the crop residues, where those residues occur in the soil, interactions of organic matter with soil minerals, and the decisions of land managers.

Under aerobic conditions, all plant components are ultimately degradable in soil and can release CO_2e. But the chemical composition of plant residues will affect the rate of decomposition. For example, perennial roots usually decompose more slowly than roots of annual crops. Deitzel et al. (2017) reported that decomposition of prairie roots was about 50% slower than that of corn roots. This effect may be related to the carbon-to-nitrogen ratio of the roots themselves, which is likelier to be lower in heavily fertilized corn systems than in perennial cropping systems. The species of perennial plants may also control their rate of decomposition. Recent laboratory incubation experiments using Iowa soils indicate that warm-season grasses may decompose faster than those of cool-season species (Ye and Hall 2020). The net contribution to soil carbon from roots in perennial systems versus the net contribution from corn-based cropping systems over several years is uncertain and demands more study.

Molecular-scale interactions of decomposed plant residues with mineral surfaces are another mechanism of persistence, especially in subsoils (i.e., below the surface layer; Mikutta et al. 2006; Wiseman and Puttman 2006). When decomposing residues are adsorbed to clay mineral surfaces or incorporated into mineral aggregates, they can be shielded from enzyme attack (Balesdent et al. 2000; Puget et al. 2000). For example, Heim and Schmidt (2007) measured lignin that was preserved for decades in the fine-silt aggregates of an agricultural soil. The persistence of plant lipids in soil, such as the suberin that occurs in roots, arises from their potential to associate with the surfaces of clay minerals as roots decompose (Huang et al. 1999; Mao et al. 2007). However, recent Iowa-based research also suggests that protection of carbon by adsorption to iron oxide minerals may be limited to well-drained soils because low-oxygen conditions in poorly drained soils lead to dissolution of the oxides and release of adsorbed carbon (Huang et al. 2020, 2021).

In many soils, a portion of organic carbon consists of charcoal residues from natural fires (Skjemstad et al. 2002). In a study of four cultivated Iowa soils, carbon in charcoal residues was shown to account for about one-third of total organic carbon (Fang et al. 2010). Charcoal carbon can be very persistent. In fact, the ages of oxidation-resistant charcoal have been estimated to be as long as thousands of years (Preston and Schmidt 2006; Liang et al. 2008). In addition, charcoal particles may be protected from decomposition when they are sequestered in microaggregates where accessibility by enzymes is limited (Brodowski et al. 2006). Modern biochar is a byproduct of industrial-scale pyrolysis of crop residues for liquid and gas products. In this process, carbon dioxide from the air is taken up by crops that are then pyrolyzed. The resulting biochar can be used as an amendment to add long-lived, persistent carbon to the soil.

Landscape position and drainage class of a soil can significantly influence the mechanisms of persistence that are described above. Across the landscape, the concentration of soil organic matter varies predictably (Schaetzl and Thompson 2015). Greater concentrations of organic carbon occur in poorly drained soils compared with well drained soils. In addition to the deposition and accumulation of eroded organic matter in low-lying positions, this can be partly attributed to the slow oxidation of carbon by anaerobic microorganisms under long periods of oxygen-limited conditions where aerobic organisms cannot thrive. The high levels of organic matter in many poorly and somewhat poorly drained soils of north-central Iowa reflect in part the drainage conditions that preceded settlement by EuroAmericans.

With these factors of soil carbon persistence in mind, land managers can increase the period of net carbon storage by practices that (1) promote deep root growth (e.g., by favoring perennial crops over annual crops), (2) promote aggregation (e.g., by limiting tillage so that root residues will be incorporated into small aggregates where they will be adsorbed to clay and isolated from enzymes), (3) limit access of oxygen to the soil (e.g., by reconstructing native wetlands where possible and by limiting new drainage of existing poorly drained soils). New research is essential to help land managers integrate such practices with one another and to predict how much carbon can be stored in Iowa soils as new practices are adopted.

4.2.3 Baseline Soil Carbon

One of the key elements of carbon markets is additionality—purchasers of credits require assurance that their investments are driving GHG emission reductions and carbon removal. A required component of assessing the level of change is establishment of defensible baselines. Implementing new agricultural practices (additionality) that can reduce GHG emissions from agriculture and other sources for extended periods of time is challenging for farmers.

Soil scientists have been inventorying the soil resources of Iowa since 1902 (Fippin 1902). As geospatial technologies have developed over time, the variety of soil properties and spatial detail in those maps has increased (Miller et al. 2019). The current, standard soil maps for Iowa were produced by the Cooperative Soil Survey, which is a collaboration between Iowa State University, county governments, the State of Iowa, and USDA-NRCS. These maps and their associated databases include the soil properties necessary for calculating soil carbon storage (e.g., organic matter concentration, thickness of zones with major differences in carbon concentration, and bulk density). However, these maps largely assume that soil properties are static. Although updates are published, they focus on attempting to improve map consistency rather than adjusting for change over time.

Modern geospatial technologies, including multi-spectral remote sensing and digital terrain analysis based on LiDAR elevation data, can now be leveraged with machine learning to map soil properties at finer spatial and temporal resolutions than traditional soil mapping. In support of soil mapping efforts, the Cooperative Soil Survey has collected soil cores from across Iowa, dating back to the 1950s. As georeferenced observations, those data points can be used to produce soil maps with the increased accuracy and precision of digital soil mapping.

A simple way to estimate the capacity of soil in Iowa to store carbon is to estimate the amount of carbon it has stored in the past. Between 1955 and 2018, over 8,000 soil cores were collected from across the state. The Geospatial Laboratory for Soil Informatics at Iowa State University is currently applying digital soil mapping techniques to construct maps of past soil properties based on those observations. Similar data are not available from before cultivation by EuroAmerican settlers began, but soil patterns from remnant prairies can be observed and extrapolated across Iowa's landscape. From these limited data, the Geospatial Laboratory for Soil Informatics has produced three maps of organic carbon stocks in Iowa soils, representing stocks before EuroAmerican settlement began in the 1800s, in the 1970s, and today.

The last of these maps is shown in Figure 4.1. Urban and subaqueous (underwater) environments were excluded because they are not included in Cooperative Soil Survey data.

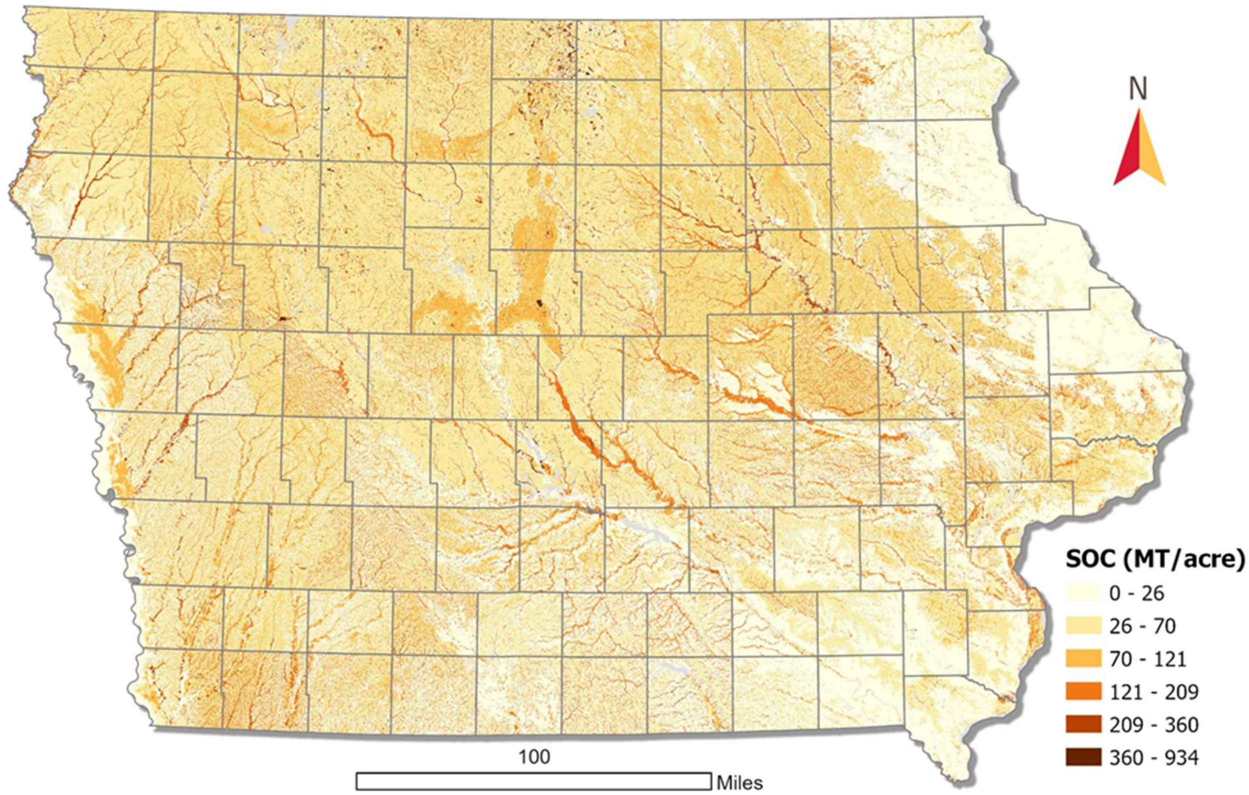

Figure 4.1. Soil organic carbon (SOC) content in Iowa soils today based on gSSURGO data (USDA-NRCS, 2021). The depth included is variable and is based on the thickness of soil with visible evidence of organic carbon accumulation (i.e., "mollic" colors, a standard for indicating soil most enriched by organic matter). For some soil areas this thickness may be up to several feet (e.g., 1–2 m).

SOC stocks are calculated by using four soil properties: bulk density, particles > 2 mm, SOC concentration, and the thickness of the soil zone of interest. Digital techniques for mapping SOC stocks have been described and evaluated by Miller et al. (2015). Spatial modelling of SOC stocks—based on the best currently available data—suggests that since the 1970s the total amount of organic carbon stored in Iowa soil has increased from approximately 950 to 990 million Mt. However, the stock of organic carbon in Iowa soils before EuroAmerican settlement is estimated to have been about 1,470 million Mt. The changes in soil carbon stocks are related to the soil's landscape position (Fig. 4.2). Although all cultivated hillslope positions appear to have lost carbon since EuroAmerican settlement, some also gained carbon since the 1970s. The largest losses are observed in the backslope and toeslope positions (Fig. 4.3). Backslopes represent the steepest slopes and are the most susceptible to erosion (Fig. 4.2). Toeslopes represent the base of slopes and are typically the wettest portions of the landscape.

While many studies have documented the accumulation of transported soil and carbon at the bases of eroded hillslopes, especially in floodplains (Knox 2006; James 2019), toeslope positions also occur in uplands that have been extensively drained. To explore the potential for changes in carbon stocks in upland toeslope positions, the Hayden Prairie in Howard County, Iowa, was studied. The distribution of carbons stocks at different landscape positions in this remnant prairie was assumed to represent topsoil thicknesses in Iowa prairies prior to widespread cultivation. The topsoil thickness observed in the footslope/toeslope position [20 in (52 cm)] was compared to the state-wide, area-weighted mean of 15 in (37 cm) for toeslope soils. Extrapolated to the entire state, the difference in thickness leads to an estimated 130 million Mt less organic carbon stored in these positions. Additional research to separate toeslopes in floodplains versus upland wetlands is needed to better understand these dynamics. One explanation for this difference could be that the aeration of soil accompanying tile drainage of toeslope wetlands led to oxidation of the organic carbon in the soil.

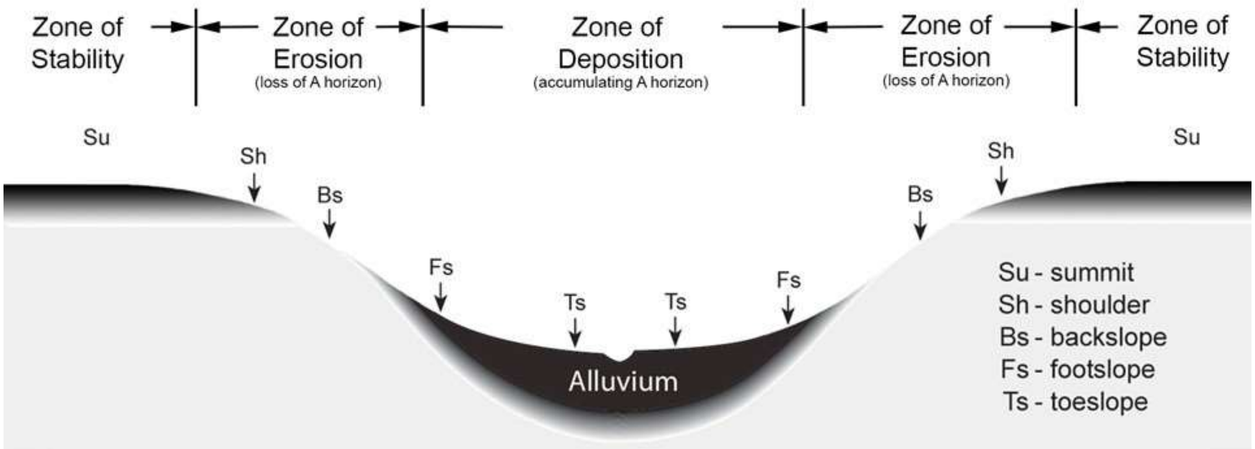

Figure 4.2. Illustration of hillslope positions with associated processes of erosion and deposition (after Ruhe 1975).

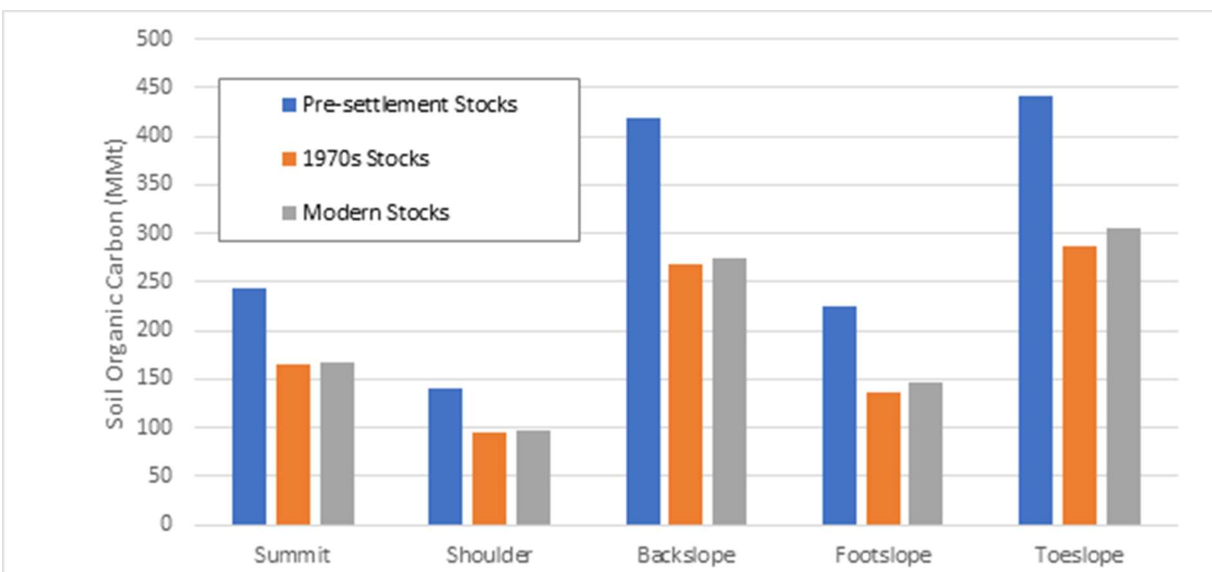

Figure 4.3. Estimates of organic carbon stored in Iowa soil by hillslope position based on historical observations and digital soil mapping (Source: Geospatial Laboratory for Soil Informatics at ISU).

Monitoring of SOC stocks should be conducted across full hillslopes because of the simultaneous effects of physical, chemical, and biological processes. As described in other sections of this report, environmental conditions and management practices may regulate biological processes that add or remove organic carbon from the soil. Changes in SOC stock at a point location could be monitored by comparing repeated samples over time. However, soil is mobile and organic carbon can move laterally, resulting in a decrease in one landscape position simultaneously with an increase in a nearby position – without changing the total amount of organic carbon stored in the soil of the hillslope. For example, if a sample point is on a backslope, management practices that incorporate crop residues could be adding carbon to the soil, but erosion of soil particles downslope could cause later sampling at the same location to show a decrease in organic carbon stored. In this scenario, the repeated sampling would appear to indicate a loss of carbon from a soil when the carbon only shifted in location. The solution to this spatial problem would be to evaluate the change in SOC stock by mapping the full area of interest (e.g., the entire cultivated field). Directly measuring soil at every hillslope position in the field would not be practical. However, the need for spatial interpolation and prediction is the regular challenge of all soil mapping efforts. While there are opportunities for improvement, soil maps, including statistical strategies to model unsampled locations to provide continuous coverage of a map area, must be relied upon for both management and policy decisions.

Strategies for providing a full accounting of SOC stock for a hillslope or field will have an effect on the calculation of SOC stock, largely with respect to resolution. A common approach to mapping soil nutrients for planning fertilizer application is to address the spatial gaps by sampling on a 2.5-ac (1 ha) grid. Soil properties at the sampling point are assumed to represent properties in the surrounding 2.5 ac (block mapping). Digital soil mapping techniques can couple machine learning with remote sensing and fewer actual samples to produce fine-resolution maps. For example, in a recent study, we intensely sampled a 36-ac (~14.4 ha) field on a 1.4-ac (0.6-ha) grid, so the map resolution was 1.4 ac. Using the block mapping approach, the organic carbon stock was calculated to be 1,065 Mt for the entire area. Using the same sample set, a digital soil mapping approach produced a map with 9.8-ft (3-m) resolution and determined the organic carbon stock to be 1,041 Mt. The coarser resolution of the block mapping approach could either over- or underestimate the SOC stock, depending on how the organic carbon is distributed within a field. These spatial issues for mapping and tracking SOC stocks illustrate the importance of accurate and consistent methodologies to support confidence from both sellers and buyers in carbon markets.

4.3 Cropping Systems and Land Management Practices

There are many options available to farmers to limit carbon and GHG emissions associated with their operations, including in-field and edge-of-field practices, and land use management. These practices may also influence other ecosystem services including soil health, nutrient cycling, water quality, and pest/disease control. Research at Iowa State University has long focused on these topics and will continue to seek to optimize these practices for profitability and environmental quality. In this section we will address the following for each set of management practices: **(1) Current science**: (a) Evidence from the scientific literature about the potential of cropping system practices to reduce GHG emissions and/or promote carbon storage, both in general and specifically in Iowa. (b) Evidence for co-benefits or tradeoffs with other ecosystem services. **(2) Science gaps**: The gaps in our knowledge that limit our ability to make specific predictions about the impacts of these practices on carbon removal. Table 4.1 summarizes the impacts of these practices on SOC and GHG emissions.

Table 4.1. Impacts of cropping systems and land management on carbon dioxide equivalent (CO_2e) in comparison to prevalent current systems of crop production. A plus (+) sign indicates likely increase in

soil organic carbon (SOC), a decrease in emissions of nitrous oxide or methane, or both. A minus (-) sign indicates likely decrease in SOC or increase in emissions of nitrous oxide or methane, or both. An equivocal (~) sign indicates directionally unclear impacts. A dot (•) indicates unknown impacts.

Practices	Management	Mitigation Outcomes (CO_2e)		
		SOC	Nitrous oxide	Methane
Artificial drainage	Standard practices	–	+	~
	Controlled drainage	~	•	•
	Shallow drainage	~	•	•
Reduced tillage	Type (e.g., no-till)	+	–	•
	Timing (fall vs. spring)	~	•	•
	Depth	+	•	•
Crop residues	Residue inputs	+	~	•
	Incorporation	~	~	•
Nitrogen fertilizer	Form (inorganic vs. organic)	~	~	•
	Optimum rate	+	+	•
	Optimum timing	+	+	•
	Optimum placement	~	+	•
Cover crops	Timing	+	~	•
	Termination method	~	~	•
	Species	~	~	•
	Duration	+	~	•
Crop rotations	Crop diversity and duration	+	+	+
Edge-of-field	Riparian buffers	+	+	~
	Saturated buffers	+	+	~
	Bioreactors	~	–	–
	Wetlands	+	~	~
Perennials	Species, harvesting, and duration	+	+	~

4.3.1 In-field Practices

4.3.1.1 Artificial Drainage

Current science. On a worldwide basis, SOC stocks have typically declined 20–40% following cultivation (Davidson and Ackerman 1993). The preponderance of loss occurs in the first 20 years of cultivation. Much of this loss can be attributed to tillage-induced erosion, aeration, and higher soil temperatures. Simulation modeling studies suggest that subsurface agricultural drainage may have also been an important cause of carbon loss in poorly drained soils (Castellano et al. 2019). Soil water content and drainage are important control mechanisms for the decomposition rate of soil organic matter and, therefore, carbon stocks in many Iowa soils, especially in the north-central part of the state. Naturally wetter cropland soils in low-lying topographic positions have higher soil carbon stocks (James and Fenton 1993) (e.g., Webster vs. Clarion soil series). However, subsurface drainage systems can release soil carbon as carbon dioxide by lowering the water table, aerating the soil, and promoting decomposition of organic matter. While there is scant research that directly relates the impacts of artificial drainage on stocks of organic carbon in soil, it is well established that draining soils increases aeration and soil temperatures, leading to increased microbial oxidization of organic carbon. Carbon markets have started to recognize the impacts: adding or improving drainage voids the carbon contract with at least one carbon company. Recently, Fernandez et al. (2017) reported a loss of more than 2.6 Mt CO_2e per ac over three years when tile drainage was installed in southern Minnesota corn-soybean research plots. Soil carbon losses of similar magnitude have been ascribed to drainage of poorly drained grassland soils in Belgium (Meersmans et al. 2009).

While subsurface drainage may promote soil carbon loss as carbon dioxide, modeling studies suggest that artificial drainage could reduce direct nitrous oxide emissions from corn cropping systems in some years and may thus lead to lower *net* cumulative GHG emissions (Castellano et al. 2019). Current research at Iowa State University will help us understand and predict how drainage practices can regulate GHG emissions that are associated with nitrogen fertilizer management.

Science gaps. The effects of artificial drainage *per se* on emissions of GHGs are often difficult to separate from the effects of other characteristics of poorly drained soils. On average, low-lying, poorly drained soils are likely to emit more nitrous oxide than strictly well-drained soils, but this may be due to greater concentrations of clay (that restricts diffusion of oxygen) or of organic matter (that supplies the mineralizable nitrogen) (Hall et al. 2018; Lawrence et al. 2021).

Still, in Iowa, the intensity and extent of subsurface drainage systems are widespread, and they could have major effects on agricultural GHG emissions. There are presently few empirical data to test this hypothesis. As drainage systems are upgraded and expanded, they can affect soil carbon stocks, carbon dioxide emissions, and nitrous oxide emissions. If Iowa is to market soil carbon-equivalent credits, we must consider and monitor agricultural drainage practices to predict how they impact both soil carbon and nitrous oxide emissions.

New drainage strategies, such as shallow and controlled drainage, could prevent major new losses of soil carbon while also improving water quality by reducing the leaching of excess nitrate-nitrogen. By draining a smaller volume of soil or retaining soil water for longer portions of the year when drains would otherwise function, these systems could accumulate and preserve larger soil carbon stocks than other practices that can also increase soil carbon, such as cover crops or reduced tillage. Simulation modeling is consistent with this hypothesis (Castellano et al. 2019), and research projects are underway to test it. Field studies that validate combinations of these management practices are also essential.

4.3.1.2 Reduced Tillage Soil Management

Current science. Tillage decreases soil density, increases soil aeration, and breaks crop residues into smaller pieces, all of which increase organic matter oxidation (decomposition) rates and loss of carbon dioxide. Global studies have consistently illustrated the negative impact of tillage on soil carbon storage and shown that reducing or eliminating tillage can result in positive carbon storage in the uppermost zones of the soil (Haddaway et al. 2017). Studies in Iowa align with these conclusions, indicating that reducing tillage will slow the rate of soil organic matter decomposition and, in many situations, result in increased soil carbon storage, especially in the uppermost 6 in (15 cm) of many soils (given that other variables are held constant). At this depth, no-till management of row crop systems in Iowa resulted in higher soil organic matter (soil carbon) contents compared to chisel plowing across five locations over a 7-year period (Al Kaisi et al. 2005). Longer-term analyses, conducted at seven Iowa sites, reinforce these findings (Al-Kaisi and Kwaw-Mensah 2020). Averaged across all locations, no-till and strip till systems added 0.11–0.13 Mt of organic carbon per ac per year for a 12-year period (0.28 to 0.33 Mt per hectare per year); chisel plowing, deep ripping, and moldboard plowing lost 0.15 Mt per ac per year (0.36 Mt per ha per year) of SOC—a difference of approximately 0.25 Mt per ac per year (0.62 Mt per ha per year). In comparison, an analysis of 351 studies conducted across the globe found that, on average, no-till stored 0.19 Mt more SOC per ac per year (0.46 Mt per ha per year) than did intensive tillage systems (Haddaway et al. 2017). Thus, Iowa research and global-scale analysis of tillage impacts on soil carbon storage align: reducing tillage improves soil carbon storage with a reasonable expectation that no-till and very limited tillage systems in Iowa will improve soil carbon stocks at the rate of approximately 0.25 Mt per ac per year (0.62 Mt per ha per year) compared to more intensive tillage systems (annual moldboard and chisel plowing and deep ripping). Reduced tillage is also associated with multiple co-benefits, such as increased water infiltration and reduced soil erosion.

Science gaps. While scientifically rigorous plot work indicates that soil carbon can be better captured with no-till row crop systems than with more aggressive tillage systems in Iowa, a quantification of the spatial variability of carbon dynamics across landscapes is absent and badly needed. Moreover, previous studies have repeatedly failed to verify or even address tillage impacts on carbon sequestration below the surface's top 6–12 in (15–30 cm) of soil. In degraded soils, such as soils on eroded side slopes, the rate of soil carbon accumulation that might occur under no-till management could be relatively large, but this research has yet to be performed in Iowa. Moving from one management system to another leads to non-linear rates of change in soil carbon. The rates of carbon change depend on the pre-existing soil carbon content, length of the period that the new system is in place, and the equilibrium soil carbon content for the new system (Guillermo et al. 2021). To identify the impact of change in tillage on soil carbon we must understand the relationships between existing soil carbon levels and equilibrium levels associated with new tillage systems across soil and climate conditions. Furthermore, information on the impact of tillage when combined with other management options, such as cover cropping, rotations for longer than two years, reduced applications of nitrogen fertilizer, and artificial drainage, is largely absent.

4.3.1.3 Crop Residue Management

Current science. Crop residues are the above- and below-ground parts of plants that are not harvested, including shoots, leaves, stems, branches, pods, flowers, cobs, brace roots, and below-ground roots. Because plant residues are composed of 40–50% carbon, their management affords some potential for carbon storage in the soil. However, leaving residues on the soil surface (as in no-till management) can lead to decomposition and loss of 70–80% of the carbon in those residues. On the other hand, surface residues protect soil carbon from loss by wind and water erosion.

How crop producers manage residues plays an important role in determining how much residue carbon, if any, will remain in the soil over extended periods. Managing crop residues for carbon storage is linked to additional benefits to crop production. Residues that are incorporated into the soil through tillage after harvest may also provide nutrients for subsequent crops, lower soil density, promote aeration, and increase both cation exchange capacity and water-holding capacity. When crop residues are left on the soil surface, they protect the surface from wind and water erosion and limit evaporation of water, providing important co-benefits of wise residue management.

There are many residue management combinations, and specific outcomes are difficult to predict in a given crop year or even over several years. Removal of annual crop residues (for example, in biofuel cropping systems) does not increase soil carbon. Karlen et al. (2011) reported that long-term stover harvest could result in a decline of soil carbon concentrations. Similarly, Laird and Chang (2013) showed that removing all above-ground biomass over a period of 19 years of a corn-soybean cropping system led to 12% less soil carbon in the upper 6 in (15 cm) of soil compared to leaving all residues in place. When crop residues are used as a surface mulch in no-till cropping systems, they normally increase soil carbon and prevent soil erosion (Ranaivoson et al. 2017).

Perennial crops, such as switchgrass, alfalfa, or trees, usually have more extensive root systems below-ground than do annual crops such as corn or soybean. They provide greater opportunity for accumulation of root residues in the soil and the accumulation of organic carbon in subsurface zones where temperatures will typically be cooler, water contents will be higher, and decomposition will be slower than near the soil surface. For example, in a study of biofuel cropping systems in central Iowa, Ibrahim et al. (2018) found that fertilized perennial biomass species in reconstructed prairies stored about 3.9 Mt more carbon per ac (9.8 Mt/ha) in the upper 12 in (30 cm) of soil than did continuous corn crops over a 7-year period. Jarchow et al. (2015) reported that reconstructed prairies in tile-drained soils that were harvested annually as biofuel feedstock produced seven times greater root mass over four years than comparable continuous corn crops.

Science gaps. While dynamic carbon models are capable of making predictions of residue decomposition dynamics, there have been few scientific studies to validate those predictions—perhaps not with the confidence needed to support financial contracts based on carbon storage. One reason is that carbon storage at soil depths below about 1 ft (30 cm) have not been prioritized, and this is where perennial crop residues are most likely to accumulate. In addition, most studies have been performed over relatively short periods (1–3 years) and do not account for long-term changes in carbon accumulation. Finally, there are many permutations of soil management practices that affect residue decomposition (including tillage, fertilization, and drainage) and a range of critical soil properties that govern decomposition process (including clay concentration, pH, and initial organic matter concentrations). Reliable, quantitative predictions for a given field will require a larger base of research data. That research should focus on refining the residue decomposition rate constants in current models and on validating carbon storage outcomes over periods as long as 10–20 years.

4.3.1.4 Nitrogen Fertilizer Management

If all other management factors, such as crop rotation and tillage, are held constant, the nitrogen fertilization rate can impact soil carbon indirectly. Nitrogen increases the biomass of the crop and therefore the amount of organic carbon fixed from carbon dioxide. Nitrogen also changes the fraction of plant carbon in above-ground versus below-ground tissues. Nitrogen fertilization further alters the carbon to nitrogen ratio of plant residues, which also regulates the rate of organic carbon decomposition (Russell et al. 2009). Studies at two locations in northern Iowa showed that with increasing nitrogen application rates on corn, above-ground organic carbon inputs (corn residue) increased while below-ground carbon was not significantly impacted (Russell et al. 2009). Organic carbon degradation rates in the soil increased in proportion to nitrogen rate such that there was generally no net change in soil carbon stocks (Russell et al. 2009). By adjusting nitrogen application rates, producers can optimize the carbon storage by maximizing below-ground residues and minimizing their rate of decomposition. Crop selection based on the amount of root growth also holds promise for increased soil carbon storage (Russell et al. 2009).

Current science. Fertilizer management practices can affect soil carbon removal by increasing plant production and thus increasing plant biomass inputs to soil carbon pools. Fertilizer management practices can also have significant impacts on nitrous oxide emissions. Across the Corn Belt and beyond, research consistently demonstrates that nitrous oxide emissions are extremely sensitive to nitrogen fertilizer rates (Shcherbak et al. 2014). Improved nitrogen fertilizer management—particularly fertilizing at a rate near the economic optimum nitrogen rate—can result in major reductions in nitrous oxide emissions while also reducing nitrate leaching and improving farmer profitability (Millar et al. 2010; Van Groenigen et al. 2010).

In Iowa corn fields, nitrous oxide emissions from soil can be the largest source of agricultural GHG emissions and can exceed potential rates of carbon dioxide removal and storage associated with other management practices such as cover crops and reduced tillage on a CO_2e basis. For example, annual nitrous oxide emissions from fertilized corn range from approximately 3 to 30 lb per ac (3.4 to 34 kg per ha) (Parkin and Kaspar 2006). However, 30 lb per ac of nitrous oxide is equivalent to 8,940 lb per ac or 4.1 Mt per ac (10 Mt per ha) of CO_2e. For perspective, this is several times more CO_2e than the carbon storage potential that might accrue when crops are raised without tillage or with cover crops (Al-Kaisi and Kwaw-Mensah 2020; see **Sections 4.3.1.2** and **4.3.1.5**). Research suggests that fertilizing at the economically optimum nitrogen fertilizer rate can reduce nitrous oxide emissions to levels below 6 lb of nitrogen per ac per year (6.7 kg per ha per year) (Millar et al. 2010). Optimum nitrogen fertilizer inputs also maximize soil carbon storage (Zhao et al. 2016; Poffenbarger et al. 2017).

Science gaps. The capacity for nitrogen fertilizer management to reduce nitrous oxide emissions in Iowa's corn-based cropping systems is poorly understood because we lack data about how nitrous oxide emissions respond to insufficient, optimum, and excessive nitrogen fertilizer rates. Current nitrogen fertilizer recommendations in Iowa provide an average fertilizer rate for all soil types and management systems (e.g., tillage, plant density and cultivar, etc.) (ISU 2021). Data to support optimization across the variables of nitrogen fertilizer rate, crop yield, seasonal weather patterns, and emissions reduction in a given cropping season currently do not exist at the scale of individual crop fields. Hence, it is a major challenge for farmers to apply the optimum fertilizer rate or to demonstrate to markets that they are indeed applying the right fertilizer rate. This is particularly important because matching nitrogen fertilizer inputs with nitrogen harvested in grain reduces both nitrous oxide emissions and nitrate leaching.

4.3.1.5 In-field Practice: Cover Crops

Current science. Cover crops, as used in Iowa, are planted after cash crop harvest and terminated before planting the next crop. Cover crop species may be planted singly, or in mixes of grasses, legumes, brassicas, and tap-rooted (e.g., tillage radish, turnip) plants. Their benefits include taking up residual soil nitrogen, slowing down runoff and erosion, combating weeds, maintaining or increasing soil carbon, and other soil health related benefits (Blanco-Canqui et al. 2015). The amount of carbon removed and stored in soils by cover crops in the short- and long-term is strongly dependent on the above- and below-ground biomass accumulated (McClelland et al. 2021). Cold winters in Iowa restrict the choice of cover crop species and the window of opportunity for plant growth and the associated benefits. The most prevalent cover crop used in Iowa is cereal rye (*Secale cereale* L.), given its winter hardiness, ability to take up residual nitrogen, and its fast growth in the spring. One option for extending the cover crop growing season is to establish the cover crop through aerial seeding (or through high-clearance equipment) before crop harvest. Another option is to delay termination in the spring, but this interferes with spring planting operations.

As long as cover crops are able to produce biomass, they are likely to increase soil carbon, but the extent of this effect depends on other factors such as tillage, soil texture and initial soil carbon, species, soil depth considered, and timeframe. Increases in soil carbon are more likely to be observed with grass versus legume cover crops, and over longer periods of time. In a global meta-analysis, Poeplau and Don (2015) estimated that cover crops can contribute about 0.14 Mt per ac per year (0.32 Mt per ha year) to soil carbon in the topsoil layer. In another study, the estimated range was 0.04–0.41 lb per ac per year (0.1–1 Mt per ha per year; Jian et al. 2020). There are many factors that can moderate the potential, and in some experiments the introduction of cover crops did not result in an increase in SOC (e.g., Tautges et al. 2019). Little change in soil carbon following cover crop adoption may be explained in some cases by increased decomposition of existing soil carbon in the presence of new carbon inputs from the cover crop (Ye and Hall 2020), i.e., a “priming effect” (Kuzyakov et al. 2000). Interactions of cover crops with other management practices may also be important. For example, cover crops’ contribution to soil carbon was higher (3.7 versus 0.5 Mt per ac; 9.4 versus 1.2 Mt per ha) under no tillage compared to moldboard plow tillage over a 12-year study in eroded soils in Illinois (Olson et al. 2014). Cover crops also contribute to soil carbon by preventing losses of soil organic matter though soil erosion (Kaspar et al. 2001; Kaspar and Singer, 2011).

Cover crops can also have an effect on nitrous oxide emissions. In a meta-analysis on the effect of cover crops on nitrous oxide emissions, Basche et al. (2014) found that, in general, the effect was neutral, but large uncertainty remains. If the goal is to reduce nitrous oxide emissions, then grasses should be favored over legumes and no incorporation of cover crop residue should be favored over tillage.

Science gaps. Most soil carbon studies have been limited to the uppermost cultivated soil horizon, the topsoil, usually between 6–12 in deep. As with other practices, when carbon deeper in the soil profile is measured, a redistribution of carbon is often observed in which gains in shallow carbon pools are offset by losses in deeper carbon pools (Tautges et al. 2019). There is a need for more long-term studies that measure soil carbon deep in the soil profile and also research that provides a full accounting of carbon and GHG dynamics associated with corn and soybean cropping systems that include a cover crop. Farmers would also benefit from additional research on managing cover crops, as the implementation of cover crops can be challenging, especially for new adopters. Timely planting and termination before cash crops is crucial. Cereal rye has, on average, a neutral effect on yield (Marcillo and Miguez 2017), but there is a real risk for yield reduction. Because biomass from corn is the main carbon input in Iowa’s dominant cropping systems, lower corn yields could reduce carbon removal from the atmosphere (Poffenbarger et al. 2017).

4.3.1.6 Extended Crop Rotation

Current science. Increasing the complexity of crop rotation systems beyond a two-year rotation of corn and soybean provides a multitude of benefits (Davis et al. 2012). The inclusion of deep-rooted perennial crops in sequences with annual crops has been suggested as a means of reducing GHG emissions and maintaining or increasing SOC stocks (Huggins et al. 1998; West and Post 2002; Russell et al. 2009). In Iowa, the use of extended crop rotations includes small grains and forage legumes in sequence with corn and soybean. Using data from a field experiment conducted in Iowa, Hunt et al. (2020) compared a conventional corn-soybean rotation with a corn-soybean-oat-alfalfa rotation using the United States Department of Energy's Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) model. The investigators found that compared to the simple 2-year rotation, the more diverse rotation used 64% less fossil energy and emitted 64% less carbon dioxide, 63% less nitrous oxide, and 76% less methane. Total GHG emissions on a CO_2e basis was 64% lower in the more diverse rotation system. The investigators attributed most of the reduction in GHG emissions in the more diverse rotation to reduced use of nitrogen fertilizer.

Results from a field experiment in southern Wisconsin reported by Sanford et al. (2012) exemplify the value of long-term investigations of soil carbon dynamics. The investigators compared changes in soil carbon stocks to a depth of 35 in (90 cm) over a 20-year period for six cropping systems, including continuous corn; corn grown in rotation with soybean; corn grown in rotations with small grains, red clover or alfalfa; and continuous grass-legume pasture. All systems lost carbon, though the pasture system almost maintained carbon stocks over the 20-year period. The investigators concluded that no-till and inclusion of perennial crops reduced SOC losses, "but neither resulted in carbon sequestration in the soil profile."

Science gaps. Existing data sets required to predict the effects of cropping systems on SOC stocks are generally incomplete. Three issues are preeminent: First, differences between cropping systems with respect to SOC stocks at a given point in time do not reveal whether there has been a net accrual or loss from an earlier date. Longitudinal (i.e., multi-year) studies are required to assess whether differences among systems reflect differences in rates of carbon loss or gain. Because soil properties, including carbon content, change relatively slowly, it may take a decade or more before differences between cropping systems are apparent. A second critically important issue is sampling depth. Accurate assessment of SOC stocks must include deep soil layers, not just the surface layer (Tautges et al. 2019). Finally, detection of differences among cropping systems regarding carbon gains and losses is made especially difficult by the large amount of spatial variation in soil carbon stocks. Kravchenko and Robertson (2011) noted that "Carbon stock measurements are often highly variable, which makes the detection of statistically significant differences among different land uses and management systems difficult. A common mistake is to interpret a lack of statistical significance as evidence for the absence of differences." Power analyses are needed to determine the appropriate number of treatment replications and soil samples for conclusions to be drawn from within experimental units. Large numbers of replicates and samples may be required to detect statistically significant differences between treatments so that the choice of one cropping system over another can be supported.

4.3.1.7 Organic Farming

Science knowns. Organic farming is a system characterized in part by the exclusion of synthetic pesticides, insecticides, and fertilizers during production (Seufert and Ramankutty 2017). The USDA has established a number of specific criteria (USDA 2021), but it also has an expansive view of organic farming as being the application of a set of cultural, biological, and mechanical practices that support the cycling of on-farm resources, promote ecological balance, and conserve biodiversity (USDA 2015). These include maintaining or enhancing soil and water quality; conserving wetlands, woodlands, and wildlife; and avoiding use of synthetic fertilizers, sewage sludge, irradiation, and genetic engineering (USDA 2015).

In organic crop production, soil fertility is maintained by applying green or composted manure, which can be sourced from within or outside the organic farming system (Porter et al. 2003). Organic farming is estimated to use 27–47% less energy than conventional systems, mostly because energy-intensive synthetic nitrogen is not used (Seufert et al. 2012; Seufert and Ramankutty 2017; Venkat 2012). Analyses have found substantial evidence for higher SOC stocks in organic agriculture than in conventional systems (Delate et al. 2015). This is attributed to the use of organic manure and compost, which provide carbon in addition to nutrient inputs (Leifeld and Fuhrer, 2010). The rate of increase in soil carbon stocks is hypothetically fastest during the transition from conventional to organic systems (Venkat 2012). Because nitrogen is the primary source of nitrous oxide emissions (Bouwman et al. 2002), the exclusion of nitrogen fertilizers leads to lower nitrous oxide emissions (Skinner et al. 2014). Organic agriculture also emits less methane than conventional agriculture (Lorenz and Lal 2016; Skinner et al. 2014).

Lower yields are the biggest drawback of organic agriculture. Food security experts maintain that high yields are essential for sustainable environmental systems because they relieve the demand for conversion of semi-natural ecosystems to agriculture (Seufert et al. 2012). On average, the exclusion of synthetic fertilizers and inefficient weed control lead to lower yields than in conventional crop production, although proper management can overcome these deficits (Delate et al. 2014; Delate et al. 2020). Including nitrogen-fixing plants in diverse rotations is crucial for garnering high yields. Some studies have found, however, that nitrous oxide emissions are higher in organic systems than in conventional farming when scaled to yield (Delate et al. 2015; Skinner et al. 2014).

Science gaps. Additional research is needed to better understand potential differences in carbon and GHG balances between organic and conventional farming. It is especially important to understand differences in nitrous oxide emissions, which are mediated by soil characteristics. A balanced picture of the role of organic farming with respect to SOC and GHG emissions should compare these variables under varying soil and climate conditions. Further research comparing organic and conventional farming on a full lifecycle basis is also needed using modelling that is integrated with field measurements.

4.3.2 Edge-of-Field Practices

4.3.2.1 Riparian Buffers

Current science. A riparian buffer is an area of trees, shrubs, and/or grasses established adjacent to streams, lakes, ponds, or wetlands. It is designed to intercept the transport of sediment, nutrients, pesticides, and other materials from nearby cropland and thereby to protect surface water. Riparian buffers provide a wide range of additional ecosystem services in agricultural landscapes, including carbon removal. In one study, SOC was shown to have increased over 8% over six growing seasons after conversion from row crops to a riparian buffer of perennial vegetation (Marquez et al. 1998). Several years of perennial vegetation was required to realize a soil carbon sequestration benefit (Marquez et al. 2017). Riparian buffers also filter and retain sediment and immobilize, store, and transform chemical inputs from agricultural uplands. Literature surveyed in the Iowa Nutrient Reduction Strategy (INRS 2017) indicated an average nitrate-nitrogen concentration reduction of 91% for water passing through a buffer root zone and an average reduction of surface runoff phosphorus concentration of 58% after establishment of riparian buffers.

One potential concern is that increased denitrification within riparian areas may result in tradeoffs between GHG (i.e., nitrous oxide) and water quality concerns (i.e., nitrate loss). Studies from Iowa and other states with similar agriculture have found that measured nitrous oxide emissions from soils within riparian buffers were significantly lower than within adjacent crop fields and that there were no observed differences in nitrous oxide emissions among different riparian buffer vegetation types (Kim et al. 2009). Additional studies in Iowa have found that re-established buffers are not sources of methane as has been found in other riparian areas or systems with more frequent saturation (Kim et al. 2010), possibly because the regional hydrology has been altered by widespread tile placement.

Science gaps. Many studies have documented increasing soil carbon stocks and reduced GHG emissions under perennial vegetation, such as comprises riparian buffers, compared to row crops (e.g., Asbjornsen et al. 2014). The rate of carbon accrual in perennial systems is often highest on degraded soils and within the first four decades, but is more subdued on higher quality soils and tapers off over time. Riparian buffer systems have been studied for 30 years, but it is not yet clear if the rate of soil carbon accrual in associated soils will slow over time.

4.3.2.2 Saturated Buffers and Bioreactors

Current science. Water-saturated riparian buffers and denitrifying bioreactors are water quality improvement practices that could have impacts on carbon and GHG dynamics. They provide substantial benefits as water quality practices, with average reductions in nitrate-nitrogen concentration of 50% and 43%, respectively (INRS 2017). The conversion of row-cropped acres to perennial vegetation for buffer establishment can furthermore remove carbon from the atmosphere as described in **Section 4.3.2.1** on riparian buffers. GHG emissions are variable, however, because sometimes denitrification is incomplete and/or methanogenesis occurs.

Saturated buffers reconnect subsurface drainage water with the soil profile to remove nitrate in tile water through microbial denitrification. Replacing cultivated land in riparian areas with a saturated riparian buffer has shown potential to remove nitrate from surface waters and to reduce nitrous oxide emissions from agricultural landscapes. Davis et al. (2019a) reported that nitrous oxide emissions from soil surfaces were greatest from fertilized corn, and that when used alone, saturated riparian buffers had significantly higher nitrous oxide emissions than traditional buffers in one out of six site-years in the study. The amount of dissolved nitrous oxide in shallow groundwater seeping from saturated riparian buffers was not significantly greater than dissolved nitrous oxide from the tile outlet. In addition, indirect nitrous oxide emissions within receiving streams were significantly reduced because of nitrate removal in the saturated riparian buffers.

A bioreactor is a trench filled with a carbon source to support microbial action through which drainage water is diverted, providing benefits similar to those of saturated buffers. Nitrous oxide is produced within bioreactors and exits through the outlet as dissolved nitrous oxide. Similarly, bioreactors can be sources of methane, with dissolved methane representing the dominant flux. While these and other studies conclude that woodchip bioreactors produce both nitrous oxide and methane, it has also been demonstrated that management strategies optimizing hydraulic residence time and mass nitrate removal can be used to limit GHG production. Similar to saturated buffers, the soil above bioreactors produced less nitrous oxide than adjacent areas under crop production. Nitrous oxide fluxes from the soil surface covering bioreactors were less than soil fluxes associated with nitrogen fertilized corn in summer months and similar to fluxes from perennial plant systems on a per acre basis (Davis et al. 2019a).

Science gaps. The literature assessing GHG emissions from saturated buffers and bioreactors is growing along with management recommendations. The majority of studies on bioreactors have been conducted over the last 10 years on bioreactors with fresh woodchips as the carbon source. Additional studies are now ongoing assessing corn cobs and corn cob-woodchip mixtures. GHG emissions from bioreactors will likely change with bioreactor age as nitrate-removal efficiency and woodchip composition change, but the rate and extent of change are yet unknown.

4.3.2.3 Wetlands

Current science. The contribution of wetlands to global warming depends primarily on the relative balance of carbon dioxide, methane, and nitrous oxide gases emitted. As a result of low organic matter decomposition rates under saturated, anoxic conditions, wetlands can be important carbon sinks and are typically much more effective than agricultural systems at sequestering carbon. However, wetlands can also be significant sources of methane and nitrous oxide (Myhre et al. 2013). The relative balance between net carbon removal and methane emission varies widely among wetlands and is sensitive to water regime. However, in terms of CO_2e, carbon dioxide removal approximately offsets methane emissions in most temperate wetlands not subject to substantial disturbance (Mitsch et al. 2013). In addition, high nitrate levels can inhibit methane production, although there is limited research on this effect in wetlands receiving agricultural nitrate loads.

Relatively few studies have quantified nitrous oxide emissions from wetlands receiving non-point source nitrogen loads. Those studies confirm that nitrous oxide emissions do increase in wetlands receiving elevated nitrate loads, but they also reveal that emission rates and emission factors are very low and that nitrous oxide flux accounts for a very small fraction of nitrogen removal (Stadmark and Leonardson 2005; Groh et al. 2015; Hernandez and Mitsch 2006; Hoglund et al. in preparation).

It is useful to note that nitrous oxide emission rates reported in wetlands receiving agricultural nitrate loads are very similar to rates for cultivated crops in the Midwest (Kim and Dale 2008; Smith et al. 2013). This is an especially important consideration in assessing the GHG contributions of wetlands used to intercept and reduce agricultural nitrate loads because it means that restoring wetlands on former cropland would not result in increased nitrous oxide emissions. At worst, emissions from such wetlands would be similar to those from the prior land use. In addition, denitrification in freshwater wetlands produces a lower fractional nitrous oxide yield than would otherwise be produced in downstream riverine and marine systems. As a result, to the extent that wetlands reduce nitrate loads to these downstream waters, overall nitrous oxide emissions across these combined systems would be reduced. These conclusions are generally applicable to water quality wetlands in any landscape position—tile-zone, break point, and flood plain wetlands including oxbows—so long as they are inundated and have sufficiently organic-rich soils.

Science gaps. Ongoing research at Iowa State University includes work on measuring and modeling nutrient removal and GHG emissions by wetlands restored for water quality improvement.

4.3.3 Land Use Practices

4.3.3.1 Perennial Crops and Reserves

Current science. Perennials are grown continuously and thus do not leave soil fallow or bare during late fall to early spring as Iowa's dominant corn and soybean annual crops do. Perennial crops include biomass crops, such as switchgrass and miscanthus, forage crops, such as alfalfa, but also novel grain crops such as Kernza®, which can produce yields for 3 to 5 years. Most of the carbon currently stored in Iowa soils originated from perennial plants—perennial grasses in particular once covered (80%) of the state (Gallant et al. 2011). Two of the main factors that affect the retention of soil carbon is the amount (and also the type) of carbon-rich organic matter that is added to the soil and the disruption of soils through tillage (see **Section 4.3.1.2** on tillage). Perennial grasses require some level of soil disruption during the initial planting year at a level comparable to that required for annual crops; disruptions in subsequent years are minimal. The lack of disruption during periods of sustained growth is a key attribute of perennials. Furthermore, unlike annual crops in Iowa, which have approximately 80 to 90% of their biomass above-ground, perennial grasses broadly allocate an equivalent amount of their biomass growth to above-ground and below-ground parts of the plant (Dohleman et al. 2009; Ordonez et al. 2020). This leaves a major fraction of annual carbon uptake to reside in the soil until it decomposes and contributes to soil organic matter.

Conversion of existing annual cropland to perennial, herbaceous cover (e.g., pasture or prairie) is a well-established practice for "marginal" or poorly performing lands. The use of perennial plant species provides many opportunities to increase soil carbon. First, living roots and above-ground biomass persist year-round and so diminish soil loss via erosion. Second, plants in these perennial systems have deeper and more expansive root systems than annual cash crops and thus increase SOC inputs. Lastly, after establishment, these perennial systems require much less management compared to cropland. Therefore, maintaining perennials for the long-term (multi-decadal scale) not only promotes SOC accrual, but also reduces emissions from fertilizer application and management operations relative to annual cropping. Strategies for carbon dioxide removal with perennial plants should also consider other, more potent GHGs such as nitrous oxide (see **Section 4.2**). Nitrous oxide emissions are an area of active research in the perennial literature, especially concerning the establishment of fertilized perennial grasses (Holder et al. 2019; Zeri et al. 2020).

Rates of soil carbon removal following establishment of perennial grasslands are highly variable but show large potential. Recently, De et al. (2020) measured SOC concentrations across multiple sites in northern Iowa and southern Minnesota where croplands had been converted to perennial grasslands (non-harvested) and maintained for 2 to ~40 years. Soil carbon removal rates were uncertain and contingent upon factors, such as topographic position, but they estimated a region-wide mean rate of 0.08 Mt carbon per ac per year (0.21 Mt C per ha per year). This rate is comparable to that estimated in a meta-analysis for annual cropland converted to perennial cropland (0–12 in or 0–30 cm; including woody crops) of 0.1 Mt carbon per ac per year (0.3 Mt C per ha per year) (Ledo et al., 2020). Further, De et al. (2020) also compiled soil carbon removal rates for perennial grasslands throughout North America and found a higher median rate for the Great Plains ecoregion (0.10 Mt carbon per ac per year, 0.26 Mt C per ha per year) than for the Eastern Temperate Forest ecoregion (0.08 Mt carbon per ac per year, 0.20 Mt carbon per ha per year). This highlights that Iowa is situated within an ecoregion more favorable to soil carbon removal through perennial establishment.

Harvest operations are a disruption that perennial croplands experience more often than the perennial conservation lands (e.g., land enrolled in the USDA Conservation Reserve Program, CRP; USDA-FSA 2021) Typically, the goal in harvesting a perennial grass is to maximize the removal of above-ground material without disturbing the soil or damaging the harvest equipment on rocks. The harvest efficiency is not likely to ever reach 100%, but it is likely that the inputs from above-ground residues to the soil surface in perennial cropping systems with biomass harvest will be lower than annual cropping systems where only the grain is harvested. In many systems, leaves can fall prior to harvest and contribute to soil organic matter.

Science gaps. Large gaps remain in how effective crop land conversion to perennial grassland is in terms of carbon removal from the atmosphere. The carbon removal rates calculated and compiled by De et al. (2020) are subject to considerable uncertainty—some rates are either indistinguishable from zero or perhaps even negative—because of the relatively low signal-to-noise of the accumulation of soil carbon. Much of this variability owes to climate and soil type, but also sampling depth. This highlights the difficulty in comparing studies and underscores the need for standardized measurements to compare changes in soil carbon stocks across sites.

There are several additional knowledge gaps when it comes to capacity of perennial systems to increase soil carbon. First, we need to improve our understanding of changes over longer timescales. While the rates reported by De et al. (2020) may be approximately linear on the decadal scale, they will eventually plateau on longer timescales (McLauchlan et al. 2006) and result in what has been called “carbon saturation,” where the carbon accumulation rate increasingly slows to no net increase. More research is needed on the long-term impacts of land conversion (i.e., >40 years).

Second, we need to improve our understanding of the permanence of soil carbon accrual. The longevity of land conversion projects, such as in the CRP, is only 10–15 years. If the contract is not renewed, typically these lands are returned to annual cropping. Considering the conversion of grassland back to annual crops, Gelfand et al. (2011) estimated a total net loss of 27 Mt CO_2e per ac (68 Mt CO_2e per ha) for CRP fields in southwest Michigan following long-term establishment of no-till corn/soybeans. Similarly, in a meta-analysis, Guo and Gifford (2002) estimated that conversion of cropland to perennial pasture increased SOC stocks overall by approximately 20%, but the reverse conversion entailed losses of greater than 50%. It is much easier to lose SOC via land conversion than it is to gain it.

Third, a fuller understanding of ecosystem service benefits and tradeoffs associated with land use conversion is needed. Perennials offer many benefits for ecosystem services in addition to soil carbon removal (Asbjornsen et al. 2014), including improved biodiversity, reduced erosion, and reduced nutrient loads to waters. Johnson et al. (2016) studied the ecosystem services provided by CRP lands in the Indian Creek watershed in eastern Iowa: the authors estimated benefits exceeding the costs paid to farmers enrolled in CRP, with a conservative estimate of $9.1 million across the 59,000-ac (24,000-ha) watershed for all services during the 10-year contract period. Perennial grasslands tend to increase soil carbon, including labile soil carbon (De et al. 2020), which may fuel the production of nitrous oxide, a much more potent GHG (**Appendix 11.2**). However, research on this potential tradeoff is scarce (Harris et al. 2015), particularly for continuous perennial grasslands. This tradeoff depends on nitrogen availability, e.g., legume species may be a potential source. Conversion of CRP grasslands to cropland with nitrogen fertilization tripled emissions relative to continuous cropland over a five-year period in one trial (Abraha et al. 2018). Although some data are available, more studies are needed to support multiple approaches to the management and valuation of perennial systems.

4.4 Research Needs

While much is already known about carbon and GHG dynamics in cropping and other land management systems, major science gaps remain that will challenge farmers' and other land managers' ability to manage those systems for carbon markets. We expect research in five overarching areas will propel the science forward: meta-analysis and long-term field experiments, which are discussed in this section; soil sampling protocols and digital soil mapping, which is covered in **Section 3.6**; process-based agroecosystem models, as discussed in **Section 9.4**; and systems research, as discussed in **Section 10.4**. Advances in cropping and land management alone are not sufficient. This research must also be well integrated with a deeper understanding of the full life cycle impacts of livestock systems, discussed in **Chapter 5**, and emerging engineering technologies related to biofuels, biochar, etc., discussed in **Chapter 6** and **Chapter 10**. The decreased emissions scenarios, such as posed in Figure 4.1 below, cannot be quantitatively evaluated without systems-level collaborative research that is interdisciplinary or transdisciplinary in character.

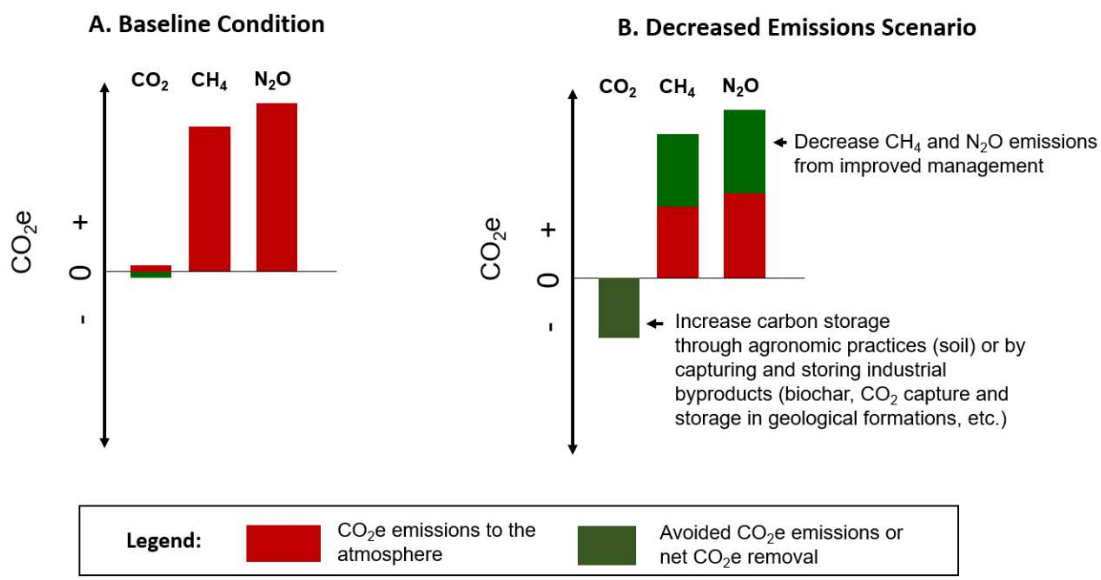

Figure 4.5. Illustration of conceptual opportunities for agricultural carbon credits under (A) expected baseline conditions and (B) a decreased emissions and enhanced carbon storage scenario. CO_2 = carbon dioxide, CH_4 = methane, N_2O = nitrous oxide, CO_2e = carbon dioxide equivalent.

4.4.1 Meta-Analysis

Perhaps the most comprehensive assessment incorporates multiple reports of a common practice (e.g., reduced tillage) and its effects on an outcome (e.g., soil carbon removal or yield) by summarizing across as many studies as possible within a given scope. This type of review is called a meta-analysis. Hence, meta-analyses are able to pool information from a variety of settings and strengthen our understanding of the key sources of variation and uncertainty. A significant caveat to meta-analyses is that, while they can provide a robust global effect estimate, they may overgeneralize and miss important contextual information. For example, Bai et al. (2019) estimated a global average increase of SOC concentration by no-tillage of 8.4%; however, this effect varies considerably with climate, soil texture, soil depth, the period since no-tillage was established. Key research questions that could be addressed by meta-analyses include:

- What are the magnitudes and uncertainties of the effects of soil and land management practices on soil carbon storage in Iowa?
- What factors (such as soil texture, climate, or hillslope position) are most important in mediating these effects?
- How comparable are the rates of potential soil carbon storage estimated using existing Iowa data versus those using soil carbon models?
- What are the co-benefits or tradeoffs for all GHGs—carbon dioxide, nitrous oxide, and methane—for each practice?

4.4.2 Long-Term Field Experiments

The classic method for observing the effect of an agricultural practice on soil carbon removal and emissions of GHGs is the plot-scale experimental field trial. Through controlled conditions, an experiment may isolate the effect of a treatment (e.g., crop rotation) on the response such as soil carbon removal. When executed effectively, such experiments provide critical insights for particular practices under controlled conditions. Therefore, plot-scale field trials often provide the primary data for meta-analyses and "ground truth" data for regional modeling studies. Drawbacks to plot-scale field studies include their cost and their limited ability to assess the temporal variability of climate and the spatial variability of soil properties. However, field experiments for cropping systems in the Midwest are fairly common. The impact of these studies on our knowledge of soil processes will be magnified when they are better supported with additional information about pertinent moderating variables (e.g., length of study, thickness of soil horizons, presence of drainage tile, seasonal weather data, prior cropping history) and making comparisons on equivalent bases (e.g., SOC stocks in an equivalent mass of soil).

Long-term field experiments at both plot scales and at field-management scales are needed to address the following questions:

- Are there interactive effects for multiple management practices (e.g., applying no-tillage and cover cropping simultaneously)? Combinations of management options are often the rule on farms and the interactions may have large effects on predicted outcomes.
- How can the spatial variability of SOC stocks and their dynamics be best characterized in the experiments?
- How can the seasonal and spatial variability of nitrous oxide emissions be best measured in the experiments?
- How can we best quantify the handling effort, measurement errors, and uncertainties in the determination of SOC stocks?
- What combinations of novel measurements and existing data can we use to better constrain the estimates of SOC stocks as well as emissions of carbon dioxide and nitrous oxide at field management scales?

- What are the long-term (i.e., decadal) effects of cropping systems management on GHG emissions, including annual, seasonal, and diurnal variations?

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Chapter 5. Livestock Practices

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5.1 Highlights

- The carbon and greenhouse gas (GHG) dynamics of Iowa animal agriculture differ significantly from national averages because of the large swine population.
- Iowa-specific data are required to determine baseline emissions and thus the extent of potential mitigation.
- Novel feed additives and anaerobic digesters are recognized as the most promising options to mitigate methane emissions of confined livestock operations.
- Increasing the use of perennial forages in diets can also help mitigate GHG emissions of cattle production.
- Holistic evaluation is required to accurately determine the true extent of mitigation provided by practices and technologies.

5.2 Background

Greenhouse gas (GHG) emissions are a natural product of animal agriculture. Of particular concern are methane (CH_4) emissions, although nitrous oxide (N_2O) emissions from manure storage and handling can be substantial. Overall, GHG emissions from agricultural activities account for about 10% of total emissions in the US, with livestock-related emissions contributing 44% to the agricultural emissions (USEPA 2021). Methane is a byproduct of animal digestion, most notably "burps" of gas from ruminant animals—including cattle, goats, and sheep. Non-ruminant animals, including swine and poultry, also produce methane through digestive processes in their intestines (as do ruminants), but at far lower rates. Methane emissions directly from animals are called "enteric." Together, enteric methane emissions plus methane and nitrous oxide emissions from manure management account for 45% of the total agricultural GHG emissions in Iowa. Roughly half of the total from livestock is enteric, and the other half from manure. In Iowa, dairy and beef still produce the largest total enteric emissions of methane, whereas the largest total emissions related to manure are associated with swine production.

Enteric emissions of methane can be influenced through genetics and adjustments to animal diet and feed additives. Some feed additives (monensin) that reduce enteric methane emissions are already widely used in Iowa cattle production. The use of these tools must also consider the impact on production (meat, milk, etc.) as well as overall profitability. Additional studies are needed.

Most cattle, swine, and poultry in Iowa are raised in confinement operations where animals are confined to roofed areas and collection and storage of manure are required before land application. Manure management systems in confined animal operations can be grouped broadly into two systems: liquid and solid manure systems. The anaerobic nature of liquid manure systems increases the potential for methane production and decreases nitrous oxide production, whereas solid manure systems can be substantial sources of nitrous oxide while contributing relatively smaller amounts of methane. The options to mitigate GHG emissions from manure include anaerobic digesters, solid separation, manure coverage, and manure additives. Although anaerobic digestion could be an effective way to reduce GHG emissions and produce energy, few farms have adopted this practice to date due to a range of economic and practical considerations. Multiple research projects are underway to examine ways to reduce these barriers.

The level of enteric methane emissions varies widely among animal species. Ruminant animals, such as cattle, sheep, and goats, are predominant methane emitters due to the large amount of methane produced in the rumen, which is 10–20 times greater than methane produced in the large intestine, also known as the hindgut (Place and Mitloehner 2010). Depending on dry matter intake (DMI) and body weight, the enteric methane emission factors vary from 120 to 280 lb per head per year (53 to 130 kg per head per year) for dairy and beef cattle, whereas the methane emission factors for hindgut emitters such as swine are much smaller at 3.3 lb per head per year (1.5 kg per head per year; IPCC 2006). Accordingly, enteric methane emissions of swine made for only 1.6% of national enteric methane emission inventories, while dairy cows contributed to a much larger percentage (24%) even though there were five times more pigs than dairy cattle in the United States in 2019 (USEPA 2021).

In Iowa, the contribution of agriculture to total GHG emissions (methane and nitrous oxide) is 31% (38 million Mt CO_2e; Fig. 1.1), which is three times higher than the contribution of agriculture to the national inventory (IDNR 2020). Methane emissions from enteric fermentation (23%) and methane and nitrous oxide emissions from manure management (22%) account for 45% of the agricultural GHG emissions in Iowa. That is similar to their contribution to the national inventory (42%; USEPA 2021). However, the contribution of manure to total agricultural emissions in Iowa is much higher than at the national level (22% versus 13%). This discrepancy could be a result of relative differences in livestock species composition between Iowa versus the national population. Nonetheless, species-specific emission data for Iowa are hard to find. Therefore, we calculated enteric and manure methane emissions of dairy and beef cattle, goat and sheep, and swine in Iowa by using corresponding emission factors (mass per animal per year) for North American livestock in IPCC (2006) and Iowa livestock population statistics (USDA 2020). Figure 5.1 presents the percentage contribution of each animal category to enteric methane (the upper panel) or methane from manure (the lower panel).

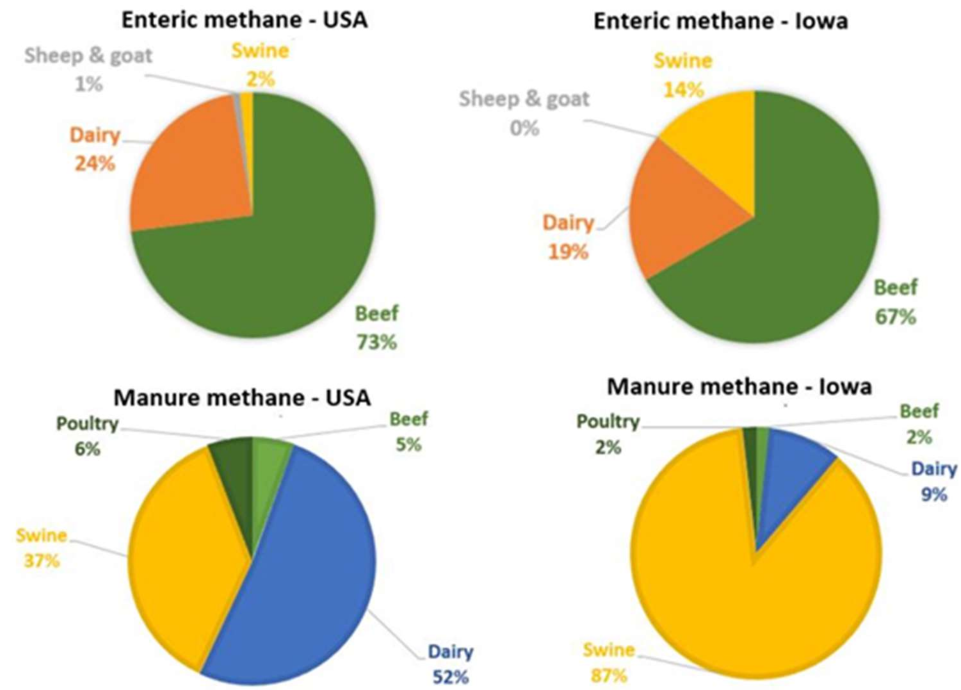

Figure 5.1. Contribution of different animal categories to enteric methane (CH_4) emissions (upper) or manure methane CH_4 emissions (lower) in Iowa and the US.

In line with the national inventory, beef cattle remain the largest contributor (67%) of enteric methane emissions in Iowa as well. On the other hand, the contribution of swine to enteric methane emissions in Iowa is seven times greater than the contribution at the national level (14 versus 2%). The contribution of swine to manure methane emissions is also more prominent in Iowa than at the national level (87 versus 37%). Dairy cattle are the largest contributor (52%) to national manure methane. Even though manure from one hog emits approximately seven times less methane than the manure from one dairy cow or heifer, Iowa's swine population being approximately 70 times greater than the dairy cattle population (24 versus 0.35 million) makes the contribution of swine about 10 times greater than that of dairy cattle in Iowa. Overall, dairy and beef cattle remain the largest source of enteric methane emissions, whereas swine is the largest contributor to manure methane emissions in Iowa. Moreover, as pointed out by Poore and Nemecek (2019), the differences between national and state emission sources highlight the need for state-specific GHG emission inventories to help develop strategies to mitigate emissions in each state.

5.3 Practices for Mitigating Methane and Nitrous Oxide Emissions in Animal Agriculture

Here, we summarize the current understanding of management practices that can be applied in Iowa to mitigate enteric methane emissions from livestock (cattle and swine), and methane and nitrous oxide emissions from their manure (the direct GHG emissions). It is noteworthy that the production of feed (e.g., grains and hay) for livestock is also associated with GHG emissions. Management practices to mitigate those indirect emissions are discussed in **Chapter 4**.

There are several options available to livestock producers to limit direct GHG emissions associated with their operations. Because enteric methane is produced during the digestion of nutrients in the diet, the options to mitigate enteric methane include dietary nutrient composition modifications and feed additives that alter the digestion process. The options to mitigate GHG emissions from manure include anaerobic digesters, solid separation, manure coverage, and manure additives. We will discuss these options below while emphasizing the evidence from the scientific literature representative of livestock production systems in Iowa. Moreover, we will discuss knowledge gaps that limit our ability to make specific predictions about the impacts of these practices on GHG emissions and carbon markets.

5.3.1 Practices to Mitigate Enteric Methane Emissions

Practices or strategies to mitigate enteric methane emissions can be classified into three categories: microbial, genetics, and nutritional. The possibility of applying vaccines to inhibit methane-producing microbiota in ruminants has been tested in few studies. The results from those studies, however, do not support a solid conclusion about the efficacy of vaccines (Baca-Gonzales et al. 2020). Genetic selection of animals for low methane production by using indicator traits, such as feed efficiency, has been researched. Results from those studies imply that more research is needed to define the relationships between methane production and feed efficiency under different production and feeding conditions (Lassen and Difford 2020). Because implementation and evaluation of nutritional strategies are much easier than genetic and microbial approaches, several have been studied comprehensively for their potential to reduce enteric methane production. We briefly discuss those approaches below. More information is available in reviews by Hristov et al. (2013), Knapp et al. (2014), Haque (2018), and Kebreab and Feng (2021).

Knapp et al. (2014) classified the nutritional strategies into three major categories: (1) controlling dry matter intake (DMI), (2) manipulation of nutrient composition of the diet, and (3) using feed additives. Decreasing DMI appears to be a promising strategy as it is the number one factor driving enteric methane production. On the other hand, DMI also drives animal production (meat and milk), and enteric methane mitigation at the expense of production would not be sustainable. Therefore, in this section we focus on modifications of diet composition and feed additives that would mitigate enteric methane emissions (in terms of mass per animal per day), while sustaining production.

5.3.1.1 Diet Composition Modifications

Diet composition can be described in terms of quantity and quality of feed ingredients (e.g., forage and grain) and nutrients (e.g., starch, fiber, and fat). Improving forage quality to achieve high digestibility and a low carbon-to-nitrogen ratio can decrease enteric methane production in cattle (Haque et al., 2018). Plant maturity significantly influences forage quality and thus enteric methane emissions. Cows consuming grass silage harvested at early heading stage produced 15% more methane than cows consuming grass silage harvested at leafy stage (Warner et al. 2017). Increasing starch at the expense of fiber can also decrease enteric methane production. For instance, through a mathematical simulation, Benchaar et al. (2001) demonstrated a 22% decline in enteric methane production of Canadian dairy cows after replacing fibrous concentrate (beet pulp) with starchy concentrate (barley grain). Hassanat et al. (2013) and Benchaar et al. (2014) showed a 6% reduction in methane emissions per pound of cow milk when alfalfa silage was completely replaced with corn silage. They also observed less organic matter in manure from cows eating corn silage, indicating a lower manure methane emission potential as well. Nonetheless, Little et al. (2017) demonstrated that alfalfa silage fields have a greater potential to store carbon in the soil than corn silage fields, and there was no difference in net carbon emissions when corn silage and alfalfa silage were evaluated on a whole farm system basis. Hassanat et al. (2017)

showed an 11% reduction in enteric methane per kilogram of cow milk when conventional corn silage in the diet was replaced with brown midrib corn silage. However, another study by the same research group showed an increased methane production from manure of brown midrib corn versus conventional corn (Benchaar and Hassanet 2019), further emphasizing the importance of evaluating the impact of dietary changes on the whole farm system basis.

Increasing dietary crude fat also referred to as ether extract within the recommended limits (< 7.0% of dry matter) has shown consistent results in decreasing enteric methane production of cattle. A systematic literature review by Eugene et al. (2008) concluded that increasing ether extract from 2.5 to 6.4% of dry matter decreased methane production of dairy cows by 9% (~1.1 oz or 30 g per cow per day). Patra (2014) showed that fatty acid composition had a marked inhibitory effect on methane production in cattle. Moreover, Jayasundara et al. (2016) reported a 0.44 oz per cow per day (13 g per cow per day) decline in methane production by Canadian dairy cows for each percentage unit increase in ether extract in the diet. The negative impact of dietary fat on enteric methane production appears to be true also for hindgut fermentation as Jørgensen et al. (2011) observed a negative relationship between dietary ether extract content and enteric methane emissions of pigs. Moreover, similar to what was observed with cattle, enteric methane of pigs was positively correlated with fiber content of their diet (Jørgensen et al. 2011). Nonetheless, further field surveys are required to collect data on baseline dietary nutrient composition of Iowa livestock to gauge accurately the impact of dietary strategies discussed above.

5.3.1.2 Feed Additives or Supplements

Feed additives or supplements capable of modifying fermentation profiles, inhibiting rate-limiting steps of methane production, or serving as alternative hydrogen sinks in the rumen have shown promising results in mitigating enteric methane emissions of both dairy and beef cattle. Those feed additives include ionophores, such as monensin, organic acids, nitrate, 3-nitrooxypropanol (3NOP), and seaweed-derived supplements (Haque 2018; Jayasundara et al. 2016; Kebreab and Feng 2021). Appuhamy et al. (2013) reported feeding monensin (32 mg per kg dry matter) decreased methane production of beef steers by 15% (19 g per animal per day) relative to the emissions of cattle not consuming monensin in the diet. The results also indicated that cattle consuming more forage and more fat in the diet could experience further reductions in methane emissions (Appuhamy et al. 2013). Monensin is well known to improve feed efficiency and decrease the incidence of ruminal acidosis, bloat, and bovine emphysema among cattle (Appuhamy et al. 2013). Almost all feedlot cattle producers feed monensin daily in Iowa. Therefore, determination of baseline enteric methane emission of feedlot cattle needs to account for the impact of monensin in the diet. On the other hand, the impact of monensin needs to be evaluated further related to the dose and nutrient composition of the diet, if they are highly variable among farms.

Feed additives, such as nitrate and 3-nitrooxypropanol (3NOP), decrease methane production by inhibiting rate-limiting steps of methanogenesis or sequestering dihydrogen that would be utilized otherwise for methane production in the rumen. Kebreab and Feng (2021) conducted a meta-analysis combining and comparing the literature data published until 2020. They found 3NOP and nitrate were the most impactful feed additives in decreasing enteric methane production of cattle. As per their results, 3NOP at 0.0020 oz per lb (127 mg per kg) dry matter decreased enteric methane production by 32 and 22% for dairy cows and beef cattle, respectively. The persistency of 3NOP in inhibiting enteric methane production is demonstrated well in Hristov et al. (2015). However, 3NOP is not used in commercial farms as it is yet to be approved by the United States Food and Drug Administration

(USFDA). Vijn et al. (2020) anticipate the incentives, such as carbon credits, would promote the use of 3NOP in the diets of commercial cattle farms once it receives the USFDA approval. Kebreab and Feng (2021) report 11.4–14.4% reductions in enteric methane of cattle for nitrate in the diet at 0.29 oz per lb (18.0 g per kg) dry matter. They also report that increasing the nitrate dose above this amount could further decrease methane production. Nonetheless, further research is required to determine the safe dose of nitrate as it can cause toxicities. The use of nitrate in ruminant diets is also not approved by the USFDA. According to Kebreab and Feng (2021), the use of only 3NOP is currently waiting for full USFDA approval.

Feed additives containing plant-derived bioactive compounds, such as tannins, saponins, and essential oils, have been reviewed for their potential to reduce methane production of ruminants (Haque 2018; Jayanegara et al. 2012; Kebreab and Feng, 2021). Moreover, these compounds are considered with increasing interest because they are perceived as "natural" (Beauchemin et al. 2008). Most of the data on the effects of tannins and saponins on methane production is, however, limited to laboratory-level experiments or animal experiments conducted in tropical countries. The experiments testing feed additives containing essential oils offer mixed results. According to a study conducted at the University of California-Davis, a feed additive based on citrus and garlic extracts containing essential oils (Mootral®) decreased methane yield of feedlot cattle by 23% without affecting DMI (Roque et al. 2019). However, a Finnish group of researchers tested the same product with dairy cows and did not see any impact on enteric methane production (Bayat et al. 2021). More data from animal experiments are required to understand the true impact of these phytogenic feed additives on methane production in different ruminant species consuming different diets under different climate conditions. Nonetheless, recently the Swiss-British Agritech Company manufacturing Mootral® launched the world's first carbon credits generated from reductions of enteric methane emissions of cattle (Mootral 2021).

In recent years, feeding seaweed supplements to cattle has attracted much attention as some experiments showed a significant reduction in methane production (e.g., 98% in Kinley et al. 2020). Lean et al. (2021) conducted a meta-analysis using data from the literature and concluded a 40% enteric methane reduction for cattle. However, the FDA limits the use of seaweed in livestock diets. Therefore, extensive research is needed to demonstrate the safety and efficacy of feeding seaweed in accordance with FDA regulations (Vijn et al. 2020). Producing enough seaweed supplements to feed livestock populations is another challenge. Vijn et al. (2020) reported that more than half of the global seaweed production is required to feed the beef cattle population in the United States alone. Moreover, the seaweed materials used in animal experiments have been harvested from the wild and it is uncertain whether commercial seaweed cultures would provide the same active ingredients. Overall, several important hurdles must be overcome before seaweed becomes a feasible option in commercial production settings.

Biochar, also known as biocarbon, is another feed supplement showing some promise in mitigating enteric methane emissions. Biochar is an extremely porous material with a high surface area that is bioactive and binds compounds containing carbon (Black et al. 2021). In an experiment conducted in Southeast Asia, biochar decreased methane production by 22% without affecting DMI of young cattle fed a low-quality diet (Leng et al. 2012). Feeding biochar, however, did not decrease the methane production of growing and finishing steers in the Midwest (Winders et al. 2019). As more data become available, the true effects of biochar on enteric methane production of cattle in the United States could be concluded. Evidence for the ability of biochar to mitigate GHG emissions from livestock manure is discussed in Section **5.3.2.7**.

Overall, there are several options to mitigate enteric methane production, measured in terms of mass per animal per day, by modifying their diet. Figure 5.2 summarizes the percentage decrease in enteric methane production of cattle for the options discussed above. Further research is needed to determine additive and interactive effects when multiples of these options are applied. For instance, it would be very important to know the net reduction of methane production when both the monensin dose and the EE concentration in the diet are increased. Further research should not be limited to only ruminant livestock. The impact of decreasing enteric methane emissions of pigs by 0.04 oz per animal per day (1.0 g per animal per day) will be similar to decreasing the methane emissions of dairy cows by 1.0 oz per animal per day (30 g per animal per day) considering the difference in the size of the average swine farm vs. the average dairy farm in Iowa. Moreover, as mentioned above, evaluating each option not only considering the enteric emissions but also the other sources of methane emissions (e.g., manure) in the farm will provide better determination of the impact.

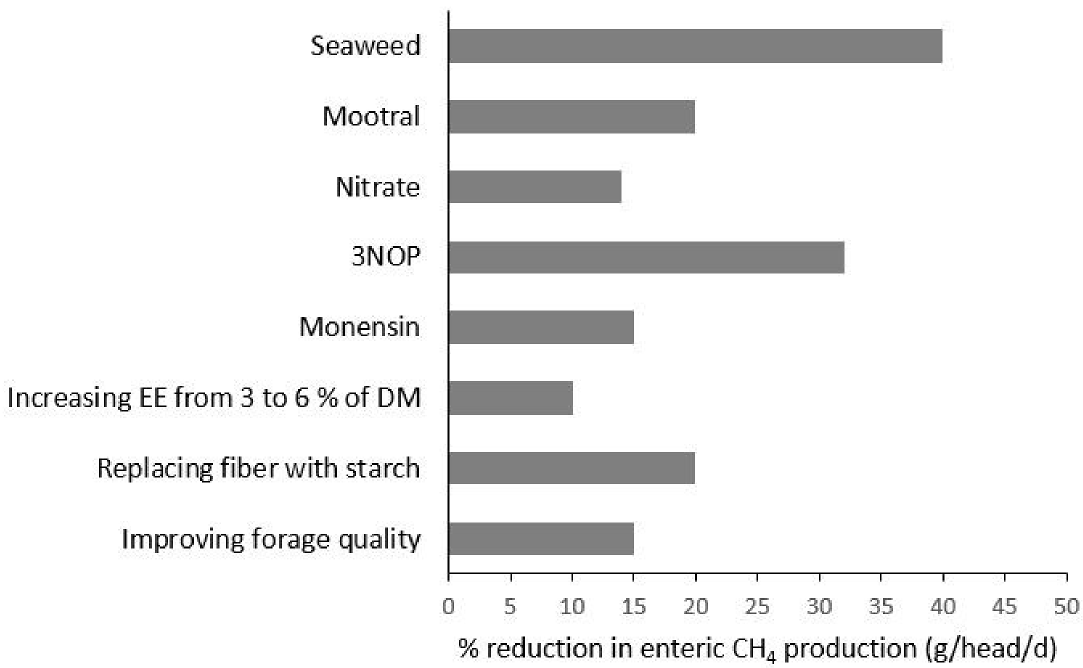

Figure 5.2. The percentage reduction of enteric methane emissions (g per animal per day) of cattle for several dietary modifications (Appuhamy et al. 2013; Benchaar et al. 2001; Eugene et al. 2008; Jayasundara et al. 2016; Kebreab and Feng 2021; Lean et al. 2021; Roque et al. 2019; Warner et al. 2017).

It is noteworthy that any practice capable of improving animal productivity (e.g., weight gain or milk yield per animal) can positively influence enteric methane mitigation. As animal productivity increases, methane intensity (enteric methane emissions per kg of meat or milk) begins to decrease allowing fewer animals to generate targeted human food production. Improving animal productivity has been the main reason behind the lower carbon footprint of the present-day cattle industry in the United States versus that of 70 years ago (e.g., 40% lower footprint of dairy industry, Capper et al. 2009). Enhancing the feed efficiency (e.g., milk yield per kilogram of feed consumed) is vital for increasing productivity and decreasing GHG emission intensities. Nonetheless, increasing feed efficiency is an ongoing effort through multiple means such as genetic selection, nutrition, and hormonal implants.

Achieving greater feed efficiencies is more challenging for ruminant livestock than nonruminant livestock (FAO 2019). The difficulty in determining the absolute nutrient (e.g., protein) requirement of ruminant animals caused by the complexities of nutrient modifications in the rumen is a major barrier to achieving high feed efficiency. Because of the high uncertainties of nutrient requirement projections, excess nutrients are fed to ensure that animals receive adequate nutrition to achieve the maximum production potential (Arriola Apelo et al. 2014). Utilizing the current knowledge about nutrient requirements (e.g., the 2021 edition of the nutrient requirements of dairy cattle by the National Research Council) will help improve feed and nutrient utilization efficiencies. Improving feed quality (e.g., organic matter digestibility) could also increase productivity and decrease nutrient excretions in manure, leading to lowered GHG emissions as well.

5.3.2 Practices to Mitigate Methane and Nitrous Oxide Emissions from Manure

The majority of cattle, swine, and poultry in Iowa are raised in confinement operations where animals are confined to roofed areas and collection and storage of manure are required before land applications. Manure management systems in confined animal operations can be grouped broadly into two systems: liquid and solid manure systems. The anaerobic nature of liquid manure systems increases the potential for methane production and decreases nitrous oxide production, whereas solid manure systems can be substantial sources of nitrous oxide while contributing relatively smaller amounts of methane (IPCC 2006).

Iowa State University conducted a survey in 2014 of current manure management practices on 22 dairy farms in northeast and southeast Iowa. The results indicated that 73% of farms had liquid manure management systems, including steel and concrete containments or an earthen lagoon (Bentley et al. 2016). On the other hand, a similar survey indicates that 99% of beef feedlot operations handle solid manure and 77% of those operations store manure in solid settling basins, confinement buildings, or both (Schulz 2014). Swine farmers use two systems to handle hog manure in barns built over slatted floors and enable urine and feces to fall through the floor into underground concrete-lined manure pits. In one system, manure in the pit is pumped out periodically into a manure spreader or other containment structure. The second system uses a pit flushing system, where water is flushed beneath the barns and directs the liquid waste to an outdoor lagoon. In the poultry industry, egg-laying hens are housed in either high-rise or manure-belt buildings. In a high-rise, manure is scraped to a storage area in the lower level of the building and is removed once a year. In manure-belt houses, manure drops onto a belt and is removed anywhere from daily to weekly to a nearby storage facility. Broilers and turkeys are raised on bedded floors. Manure mixes with the bedding, forming litter, which is removed to storage or land applied after one or more flocks have been produced (CALS 2005).

We discuss some important practices that can mitigate methane and nitrous oxide emissions from manure management systems in the following sections. Overall, current literature indicates that there are a number of options available to mitigate GHG emissions from animal farm manure. However, the efficacy of each option depends on many factors, including the type of manure storage, the amount and composition of manure and other materials going into the storage, and weather conditions. Moreover, it is noteworthy that the land application methods can significantly influence the overall GHG emissions from manure in a given farm. For instance, using 75 observations (69 from field experiments), Hou et al. (2015) showed that nitrous oxide emissions from injected manure is nearly two times greater than the emissions from surface broadcasted manure.[14] Figure 5.3 presents the impact of some manure storage and land application practices, alone or in combination, on GHG emissions from slurry manure relative to a reference. The reference included a conventional dietary crude protein content, an untreated manure storage without a crust formation, and surface-broadcasted manure. The "diet" represents a 2-percentage unit reduction from the conventional crude protein content. Additional details of these practices are also available from Iowa State University (2021).

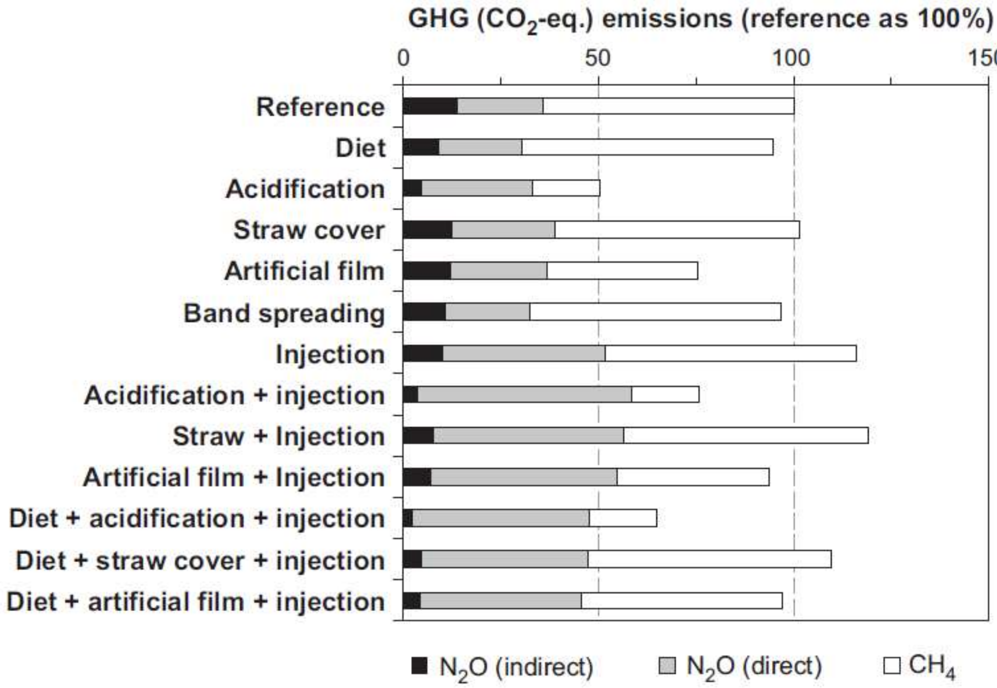

Figure 5.3. The individual or combined effects of manure storage and land application practices on GHG emissions from slurry manure relative to a reference (source: Hou et al. 2015).

5.3.2.1 Anaerobic Digestion

Anaerobic digestion is a process that promotes the degradation of organic matter in biological materials in an oxygen-free environment using bacteria (Massé et al. 2011). This process results in biogas and a nutrient-rich, relatively stable, and odorless sludge. Biogas contains 60–70% methane (Massé et al. 2011) that can be captured and used as a renewable fuel. When designed and operated properly without gas leakages and emissions from the sludge, anaerobic digestion offers one of the most effective methods for reducing methane emissions from manure (Frear et al. 2011). The establishment of anerobic digestion is easier for farms already having liquid manure systems than solid manure systems (Jayasundara et al. 2016). Moreover, anaerobic digestion systems are effective in generating carbon credits because the baseline GHG emissions for liquid manure systems is significantly higher than for solid manure systems (IPCC 2006; Jayasundara et al. 2016). As an example, anaerobic digestion

[14] As with many practices, there are tradeoffs—surface application of manure without incorporation can be a water quality concern.

decreased manure methane emissions in a dairy farm in the United States with 6200 cows by 47% compared with emissions from a reference system (Artrip et al. 2013). Although anaerobic digestion could be an effective way to reduce GHG emissions, only few farms have adapted it because of the high capital cost for digester construction, limited competitiveness of biogas with other fuels, and insufficient waste (Frear et al. 2011). According to current data from the Iowa Department of Natural Resources (IDNR; see also **Chapter 7**), there are 12 anaerobic digesters in Iowa. The data also shows that Iowa has 8769, 743, 206, and 271 swine, beef, dairy, and poultry facilities with ≥ 300 animal units, respectively, indicating the possibility to promote more anaerobic digesters in the state. A feasibility assessment for Minnesota farms by Ciborowski (2001) pointed out that with a tax incentive of $0.015 per kWh-generated, anaerobic digestion on dairy farms with as few as 250 to 300 cows could be economically feasible. Some farms co-digest animal manure with other materials such as food processing wastes, slaughterhouse wastes, and other feedstock (USEPA 2001). The co-digestion will increase the carbon-to-nitrogen ratio and thus methane production potential of the digestate. However, co-digestion can also increase methane emissions from the sludge or the effluent. Therefore, future research measuring methane emissions both during and after anaerobic digestion will help with the correct determination of the reduction of methane emissions to the environment.

5.3.2.2 Composting

Composting is an aerobic process that transforms biological waste into a stable, humus-like material (Brown et al. 2008). The composting process can be "active" aeration supplied by frequent turning, or "passive," with only natural aeration (Peigne and Girardin 2004). However, stockpiled manure is not considered composting. Jayasundara et al. (2016) compared methane and nitrous oxide emissions from composted vs. stockpiled dairy cow manure across four independent studies and reported up to 80% reduction in GHG for composting under warm weather conditions. They also showed that cold and wet conditions adversely affect composting and could even increase GHG emissions. Multiple factors, including ambient temperature and humidity, affect the impact of composting on GHG emission mitigation and are responsible for the inconsistent results in the literature (Hou et al. 2015)

5.3.2.3 Complete Emptying of Manure Storage

The old manure left in the storage can act as an inoculum, facilitating the rapid reestablishment of microbes producing methane. Massé et al. (2008) reported 2–5 month lag phases of low methane emissions when manure was loaded into clean storage vessels. Wood et al. (2014) evaluated the impact of complete versus partial (50%) emptying of dairy cow manure (liquid) in Canada, and observed about 50% lower total GHG emissions (methane and nitrous oxide) from the completely emptied manure storage compared with emissions from the partially emptied storage. Nonetheless, Iowa-specific data on the degree of emptying manure storage and related emission measurements are required to establish baseline emissions to evaluate the GHG reductions in separate livestock categories.

5.3.3.4 Solid-Liquid Separation

Solid-liquid separation is a process that results in a solid fraction that is rich in dry matter and nutrients, and a liquid fraction relatively low in both dry matter and nutrients. Several different systems for separating manure slurry into solid and liquid fractions are available, and their performance under practical farm conditions has been reviewed by Hjorth et al. (2010). However, nitrous oxide emissions during the storage of separated solid and liquid fractions can increase significantly because of the existence of aerobic/anaerobic zones in the stored solids, similar to conditions in solid manure piles

(Fangueiro et al. 2008a). On the other hand, methane emissions from the two fractions can decrease significantly. Because methane constitutes nearly 90% of the total GHG (by CO_2e), Fangueiro et al. (2008a) still observed a net decrease of about 23% of the total GHG (nitrous oxide + methane) emissions from solid and liquid fractions compared to the emissions of unseparated manure. Amon et al. (2006) and Fangueiro et al. (2008b) also observed increased nitrous oxide emissions and decreased methane emissions from liquid and solid fractions applied on crop fields. Again, the net GHG emissions from separated solid and liquid fractions were still 20% to 37% lower than that from unseparated manure.

5.3.2.5 Covering Slurry Storage

The coverings can be either fixed structures or floating material to create a barrier over the free surface of the manure. Covering can take two forms, natural and artificial. Natural covers can be the “crust” that formed naturally on the surface of liquid manure or a layer of natural material (e.g., straw or wood chips) added to the surface of the storage. Among artificial covers, plastic covers are the most widely used. Covering was first recognized as an effective method to mitigate ammonia emissions. However, later experiments indicated appreciable reductions (up to 26%) of methane and nitrous oxide (VanderZaag et al. 2008). Hou et al. (2015) reported an increase in nitrous oxide emissions from straw-covered manure, but found a 98% reduction in nitrous oxide emissions when the slurry was covered with artificial films.

5.3.2.6 Manure Acidification

Acidification of poultry litters has become a more common practice than acidification of slurry manure in Iowa. The acidifying strategies include dietary modifications (e.g., adding phosphoric acid or fiber to the diets) and dosing the manure with strong acids (e.g., sulfuric acid) or weaker fermentable acids (e.g., lactic acid). Nonetheless, adding acids to manure has shown greater reductions in GHG emissions than dietary modifications. Reviewing the results of five studies (one pilot-scale and four laboratory-scale), Huo et al. (2015) demonstrated that acidification of slurry manure decreased the methane emissions by 87%. Analyzing a larger data set retrieved from the literature, Kebreab and Feng (2021) reported a 59% decrease in methane emission from manure after acidification. More information on the relationships between acid dosage, manure type and composition, and environmental factors from field-scale trials is required to develop indices to accurately determine the changes in GHG emissions caused by acidification and other mitigation strategies.

5.3.2.7 Manure Additives

There are many different types of additives, particularly for anaerobic storages such as liquid or slurry. The farm operators use these products to reduce odors, enhance manure homogeneity, decrease crust formation, and decrease nutrient losses. Although these additives do not claim to decrease GHG, some hypothesize that additives can have this effect (Cluett et al. 2020). A group of researchers at Iowa State University evaluated 12 manure additives on the market for their ability to mitigate methane and nitrous oxide emissions from deep-pit swine manure (Chen et al. 2020). None of the tested products showed a significant reduction in GHG emissions. Results to date on the effect of the addition of biochar on methane emissions are contradictory (e.g., Maurer et al. 2017; Kebreab and Feng 2021), suggesting the need for further research.

5.3.2.8 Diet Composition Modifications

Changing dietary nutrient composition can change gaseous emissions from manure. Decreasing dietary crude protein has been shown to significantly decrease ammonia emissions from manure (Hou et al. 2015). Decreasing ammonia emissions can mitigate indirectly the production of nitrous oxide in the wider environment outside the farm (IPCC 2006). Additionally, decreasing dietary crude protein can decrease nitrogen in feces and urine and thereby result in lower nitrous oxide emissions from manure (IPCC 2006). Any manipulation of dietary crude protein must make sure that essential amino acid requirements for meat, milk, and egg production are met. Dietary crude protein in swine and poultry diets can be decreased without scarifying meat or egg production by adding crystalline essential amino acids to the diet. Because the microorganisms in the rumen degrade crystalline amino acids, they have to be protected from rumen degradation before being added to cattle diets. The impact of lowering crude protein alone or with essential amino acids (crystalline or rumen-protected) on manure GHG emissions is poorly understood. Feed additives that decrease enteric methane emissions are hypothesized to have a residual effect on methane emissions from manure. However, current literature does not support this hypothesis (Owens et al. 2020). On the other hand, some data highlight increased manure emissions caused by feed additives applied to mitigate enteric emissions (Lucas et al. 2021).

5.4 Greenhouse Gas Emissions from Grazing Cattle

Because of their capacity to photosynthetically sequester atmospheric carbon into organic carbon and reduce soil erosion, pastures containing perennial forages have the potential to act as a sink for atmospheric carbon (Soussana et al. 2007; Franzluebbers 2005; Liebig et al. 2010; Teague et al. 2016), thus maintaining or increasing soil organic carbon (SOC) stocks by more than 6 to 17 times of those in row crop agricultural systems (Sanford et al. 2012). However, the capacity of grasslands to sequester SOC is influenced by climatic conditions such as temperature and precipitation, soil texture, moisture, and fertility, pasture age and plant species, and agricultural management (Liebig et al. 2010; Sousanna et al. 2007; McSherry and Ritchie 2013), making quantification of the potential to sequester carbon in pasture soils difficult. However, these factors also provide opportunities to increase SOC through management. For instance, nitrogen fertilization will increase SOC sequestration as plant biomass in pastures (Allard et al. 2007; Liebig et al. 2010). Animal grazing may increase SOC sequestration in comparison to ungrazed grasslands (Franzluebbers 2005), but grazing did not affect or even decrease SOC in other studies under rangeland soil and climatic conditions (Milchunas and Laurenroth 1993; Derner et al. 1997). In addition to climatic conditions, soil, and plant species, grazing management may contribute to the variation in these results. Increasing stocking rate in a continuous stocking system increased SOC accrual in a long-term study (Liebig et al. 2010), but this may have resulted from a change in forage species. However, excessive grazing intensity may reduce SOC sequestration by limiting photosynthetic tissue and subsequent root growth (McSherry and Ritchie 2013). Grazing by rotational stocking increased total SOC by 22% in comparison to pastures grazed by continuous stocking or harvested for hay (Conant et al. 2003). Similarly, grazing by rotational stocking at relatively high stocking rates increased SOC sequestration in comparison with continuous stocking at an equivalent stocking rate (number of animals per ac) (Teague et al. 2016) or mob stocking at a lower stocking rate but greater stocking density (lb of animals per ac) (Chiavegato et al. 2015a). However, while SOC distribution changed, total SOC content did not differ across long-term (50+ years old) grass pastures grazed by continuous, strip, or rotational stocking over only three years in southern Iowa (Russell 2019).

While the multiple factors affecting soil carbon sequestration in pastures provide challenges in the quantification of their role in mitigating climate change, these challenges are exacerbated by the emissions of enteric methane by grazing livestock and the oxidation or emissions of carbon dioxide, methane, or nitrous oxide from the soil which offset atmospheric carbon sink potential of pastures by 19 to 89% (Allard et al. 2007; Soussana et al. 2007). Pasture management practices that reduce fiber concentration and increase dry matter digestibility may reduce enteric methane production per unit of DMI. Thus, beef cows grazing alfalfa-grass pastures produced 25% less methane per unit of energy intake than cows grazing grass pastures (McCaughey et al. 1999). Similarly, grazing by rotational stocking may reduce total methane emissions compared to grazing by continuous stocking of warm season grasses (DeRamus et al. 2003). However, more commonly, when pasture forage is enhanced through such management practices as increased stocking rate or rotational grazing, the factors reducing methane production per unit of DMI require more forage, thereby resulting in no differences in daily enteric methane emissions by the cattle in different grazing systems (McCaughey et al. 1999; Pinares-Patino et al. 2007; Chiavegato et al. 2015b; Russell 2019). These results imply that methane production from cattle grazing in systems that improve forage quality, such as grazing legume species by rotational stocking, may be reduced by use of cattle genetically selected for improved feed efficiency (Hegarty et al. 2007). Pasture soils may serve as a sink or source of methane, depending on the balance between methane-consuming and methane-producing microbial communities (Chiavegato et al. 2015c). Methane sink activity seems to be greater at lower soil moisture contents (Liebig et al. 2010), but soil oxidation or emissions of methane have not been affected by the presence or absence of grazing or grazing management practices (Allard et al. 2007; Liebig et al. 2010; Chiavegato et al. 2015c).

Pasture soils are most commonly sources of nitrous oxide with their emissions being greatest in moist soils and increasing with nitrogen fertilization (Allard et al. 2007; Liebig et al. 2010). Thus, soil nitrous oxide emissions may reduce some of the effects of increased carbon sequestration resulting from nitrogen fertilization (Allard et al. 2007; Liebig et al. 2010). There is no clear effect of grazing or grazing systems on nitrous oxide fluxes (Chiavegato et al. 2015c), which may be the result of temporal-spatial variations associated with soil, climate, and fecal and urinary excretion across pastures (Flechard et al. 2007; Soussana et al. 2007).

Grazed grasslands provide the potential to sequester large quantities of atmospheric carbon. In addition, regenerative grazing of these grasslands can mitigate water pollution and other economic and social problems associated with row crop agriculture in the upper Midwest (Spratt et al. 2021). Unfortunately, 58% of the grasslands present in the Midwest have been lost to other land uses between 1945 and 2017, with a third of that loss occurring in the decade before 2017 (Russell et al. 2018). However, the amount of carbon that may be sequestered is subject to considerable variation associated with soil, climate, plant, and management factors. While regenerative grazing is promoted as a system using a high stocking density with long rest periods to enhance SOC (Teague et al. 2016), the optimum grazing system in any given pasture will require site-specific management. The optimum system, as related to the length of grazing and rest periods and stocking density, needed to maximize carbon sequestration will depend on climate, soil, and plant conditions within a pasture. Furthermore, while sequestration of carbon dioxide by plants should have the largest effect on global warming, emissions or uptake of methane and nitrous oxide may offset some of the benefits of SOC on GWP. Fluxes of these gases vary with animal nutrition and genetics as well as climatic and soil conditions. Thus, to truly be able to quantify the ability of grazed grasslands to mitigate global warming, considerable research in different locations with interdisciplinary teams of soil scientists, plant agronomists, ecologists, animal scientists, chemists, and agricultural engineers is needed.

5.5 Modeling Tools for Livestock Systems

The following models can help evaluate the aforementioned GHG mitigation strategies, mostly on a whole-farm-system basis.

5.5.1 AMPAT

The Air Management Practices Assessment Tool (AMPAT) is web-based and available at no charge (ISU 2021). The purpose of the AMPAT is to provide an objective overview of mitigation practices best suited to address odor and gaseous and particulate matter (PM) emissions at livestock operations so that different mitigation techniques can be compared effectively. Practices are divided into three categories based on emission source, including animal housing, manure storage and handling, and land application. For each emission source, the comparison includes a summary page that provides quick visual assessment of mitigation performance for various parameters including GHG emissions. Each mitigation practice then has an individual page, which includes a printable fact sheet; a short online slide presentation; a conservative estimate of the range in effectiveness for mitigating ammonia, PM, odor, volatile organic compounds, and GHG; and a relative cost.

5.5.2 IFSM

The integrated farm system model (IFSM 2021) is a process-based simulation model that was originally developed by linking alfalfa and corn production models with a dairy animal intake model to predict on-farm feed production and use. This model was then extended to a full dairy farm model by adding components for simulating feed storage, animal performance, manure handling, tillage and planting operations, environmental conditions, and machinery use (see Figure 5.4). Unlike many other farm-scale models, IFSM simulates all major farm components at a process level, allowing integrated assessment of agro-environmental implications (carbon, nitrogen, and phosphorus transformations and losses) and economic implications of specific management practices at the farm scale. Recently, IFSM has been used for environmental assessment of various dairy farming systems in the US. The IFSM model functions on all major Windows operating systems. Input information is supplied to the program through three parameter files. The farm parameter file contains data describing the farm such as crop areas, soil type, equipment and structures used, numbers of animals at various ages, harvest, tillage, and manure handling strategies, and prices for various farm inputs and outputs. Simulation output is available in four files that contain summary tables, report tables, optional tables, and parameter tables. The summary tables provide average performance, environmental impact, costs, and returns for the years simulated. These values consist of crop yields, feeds produced, feeds bought and sold, manure produced, nutrient losses to the environment, production costs, income from products sold, and the net return or profitability of the farm.

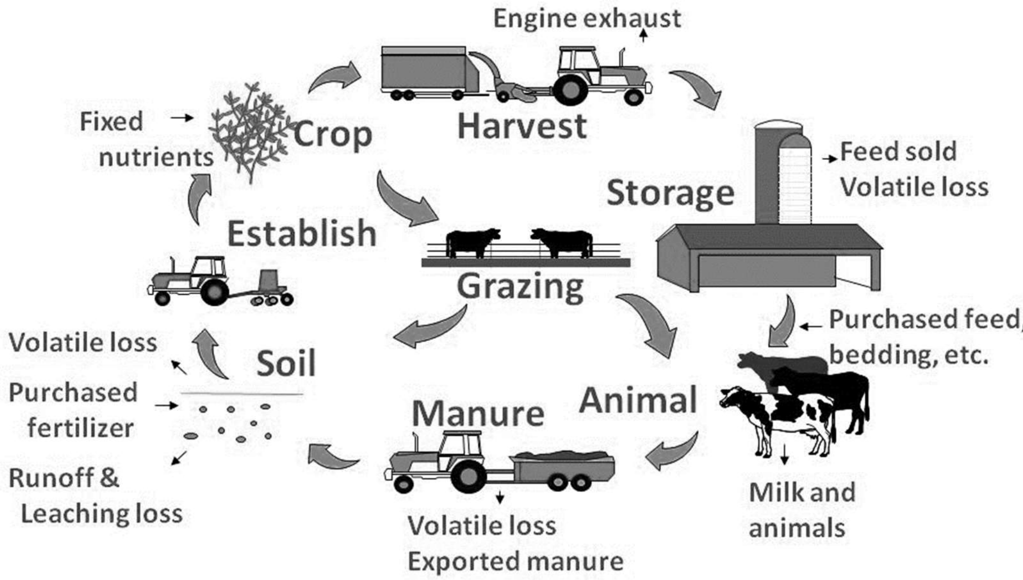

Figure 5.4. The carbon flows represented in the integrated farm system model (IFSM).

5.5.3 Manure-DNDC

The relationships between the environmental factors, the reactions, and the gas production have been incorporated in the process-based DeNitrification-DeComposition (DNDC) model to describe manure organic matter turnover and gas emissions. Using Manure-DNDC, the users can construct a virtual farm by selecting and integrating one or more of the candidate farm facilities (i.e., feedlot, compost, lagoon, anaerobic digester, and cropping field) parameterized in the model. Manure-DNDC calculates variations of the environmental factors for each component facility based on its technical specifications, and then utilizes the environmental factors to drive the biogeochemical reactions (Li et al. 2012).

5.6 Research Needs

Most of the tools available to measure enteric methane emissions are for research purposes (see **Appendix 11.4**). While there are several empirical models to predict enteric methane emissions from livestock in Iowa, the ability to determine dry matter intake (DMI) is critical for achieving accurate and efficient predictions from those models. In addition to DMI, both enteric and manure methane emissions could be variable depending on diet composition. Moreover, the efficacy of some mitigation practices discussed below can vary depending on diet composition. Methane production and DMI measurements, and diet composition data representative of Iowa livestock are needed to evaluate models for their predictive power and practices for their mitigation efficacy.

Researchers measuring methane emission in commercial operations have found significant variations in emission amounts at the animal housing level (Haeussermann et al. 2006), caused in part by broad diurnal and seasonal variation (Hartung 1998). Methane or nitrous oxide emission factors will invariably have significant uncertainties when applied to specific farms. Consequently, many livestock operations need to be surveyed for greater confidence in the emission factors used, which may not be practical given the cost of performing these studies. While models and estimates based on standard reference sources are also widely used, standard references and models may not represent Iowa conditions generally, or important farm to farm differences in Iowa. More Iowa data are needed.

There are a number of key science gaps in our understanding of GHG emissions in grazing systems, some of which are common to overall animal agriculture:

- The effects of the duration of forage stands of different species to sequester carbon dioxide as SOC.
- The interaction between the stocking density of grazing systems as it relates to herbage removal and length of rest periods on SOC content.
- The trade-off between SOC storage with enteric methane production and soil carbon dioxide, methane, and nitrous oxide fluxes in pastures grazed at different stocking densities in different grazing systems.
- Quantification of the potential of cattle genetically selected for improved feed efficiency as defined by residual feed intake to reduce methane production by grazing cattle.
- Quantification of the potential of nitrification inhibitors to reduce pasture nitrous oxide emissions.

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Chapter 6. Agriculturally Based Engineering Technologies

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6.1 Highlights

- There are many opportunities for engineering technologies to contribute to reducing greenhouse gas (GHG) emissions and carbon removal, including those that are agriculturally based.
- Iowa could produce a wider range of energy carriers suitable for reducing GHG emissions than it already does.
- Iowa could become a leader in carbon removal and facilitate the transition to a low-carbon economy by processing agricultural biomass into advanced biofuels and materials that sequester carbon.
- Research and demonstration projects—conducted in realistic environments and integrated with agricultural systems—are needed to evaluate the technical and economic viability and potential benefits and costs of engineering technologies.

6.2 Background

Iowa has the opportunity to contribute to decarbonization through transportation, power, heating, and chemical manufacturing. A multitude of low, zero, and net negative carbon technologies have been or are being developed. Many different factors will influence which technologies win out. For example, technology readiness, the ability to leverage existing resources, projected return on investment and financial risk, prospects for scaling, impacts on society and the environment, public acceptance, and public policy are all likely to come into play.

This chapter discusses engineering technologies that hold promise for Iowa, and related factors. Approaches that can either reduce the rate of GHG emissions, remove atmospheric carbon dioxide, or contribute to both are considered. While reducing GHG emissions can slow and even stop the rise in atmospheric GHG concentrations, only carbon removal strategies can lower atmospheric carbon dioxide concentration (NASEM 2018). For this reason, and because Iowa is a leading agricultural state, agriculturally based engineering technologies receive much of the focus. However, a complete view of these technologies considers connections with the electric grid.

Simultaneously decarbonizing agriculture and the electrical grid provides opportunities for economic synergies among the two systems. For example, resources associated with the electrical grid may be powered by multiple fuels (wind, solar, renewable fuels). Low-cost electricity can be stored in batteries, but also as liquid fuels, some of which could be carbon zero (e.g., ammonia, hydrogen) and others which have the potential to be carbon negative (e.g., ethanol, biodiesel, and renewable natural gas). Such synergies may lead to greater economic development in Iowa, greater resiliency for the electrical grid, and thus potentially lower costs for energy rate payers. The key will be the development of flexible, scalable, interoperable systems—an area in which Iowa is poised for leadership.

6.3 Opportunities for Reducing Greenhouse Gas Emissions

Most GHG emissions associated with residential, commercial, transportation, and industrial sectors of the economy are the result of energy generation from fossil fuels. Some come from chemical transformation of materials into products, most prominently manufacture of steel, concrete, and nitrogen fertilizer. Substitution of fossil fuels by renewable energy can achieve significant GHG emission reductions across these sectors. Electrification via renewable power, such as wind and solar energy, has received much attention in this respect and notable progress has been achieved to date.

The enormity of the task of moving to a zero-carbon economy cannot be understated, requiring rapid development of enabling technologies, massive investment in new infrastructure, and shifts in consumer acceptance of a wide range of electrified products, ranging from battery-powered electric vehicles to induction cooktops. Transition to a low-carbon economy without unduly disrupting the lives and livelihoods of Iowa's citizens will best be accomplished by the measured introduction of new technologies and infrastructure while simultaneously adopting "drop-in" substitutes for fossil fuels that are compatible with existing infrastructure and can quickly contribute to GHG emissions reduction goals. For example, battery-powered electric vehicles currently represent only 4% of sales of passenger vehicles and 1% of vans and trucks (Bloomberg NEF 2021). A recent United States Energy Information Administration (EIA) report projects that spark ignition engines will dominate light duty vehicles (>70%) as late as 2050 (Fig. 6.1). Clearly, low-carbon fuels compatible with spark ignition engines will be important to achieving significant reductions in carbon emissions for decades to come (Farrell et al. 2020). Most of these drop-in substitutes are advanced biofuels manufactured from agricultural and forestry residues or dedicated energy crops.

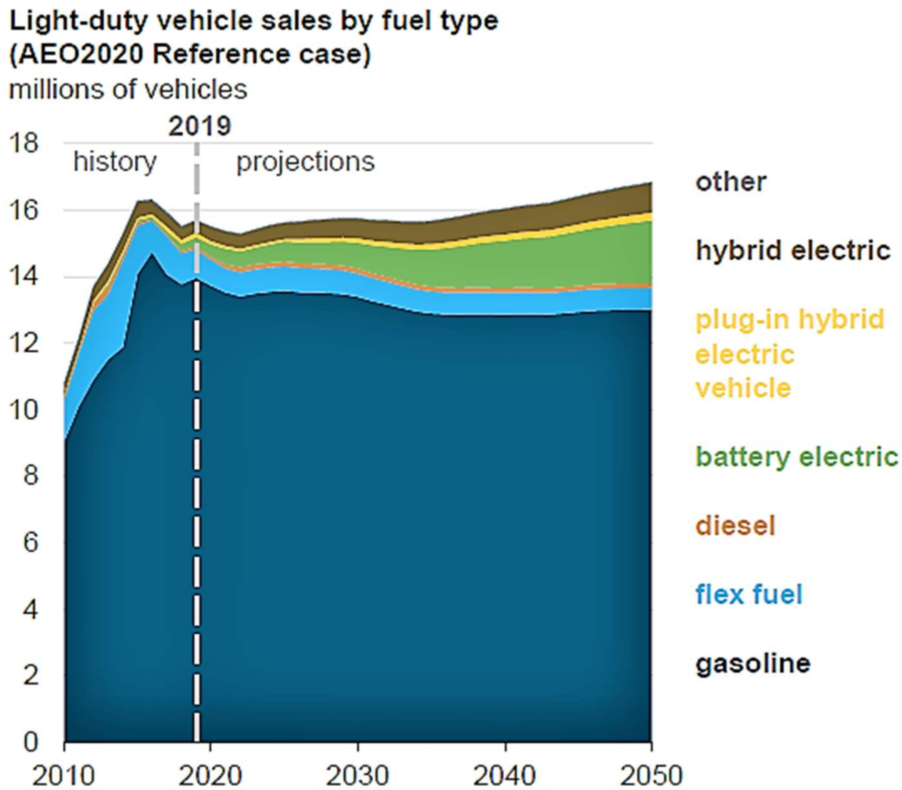

Figure 6.1. Projections of light duty vehicles by class through 2050 (USEIA 2020a).

This section of the report explores opportunities for GHG emissions reductions in various sectors of the economy and their low-carbon energy carriers. Table 6.1 summarizes several categories of opportunities to reduce GHG emissions and suitable low-carbon energy carriers for each. Each of these opportunities is presented in detail, followed by descriptions of the various low-carbon energy carriers suitable for use in these opportunities.

Table 6.1. Opportunities for reducing greenhouse gas (GHG) emissions and suitable low-carbon or zero-carbon energy carriers relevant to Iowa.

Opportunity for reducing GHG emissions	Suitable Low-Carbon or Zero-Carbon Energy Carriers
Light Duty Vehicles	Hydrogen, renewable natural gas, high-octane low-carbon fuels, ethanol
Heavy Duty Vehicles	Hydrogen, renewable natural gas, ethanol, biodiesel, renewable diesel
Aviation	Sustainable aviation fuel
Maritime Transport	Hydrogen, green ammonia, renewable fuel oil
Dispatchable Renewable Power	Hydrogen, renewable natural gas, green ammonia, biodiesel, renewable diesel, renewable fuel oil
Process and Residential Heating	Hydrogen, renewable natural gas, green ammonia, biodiesel, renewable diesel, renewable fuel oil

Chemical Manufacturing	Hydrogen, renewable natural gas, green ammonia

6.3.1 Opportunities for Reducing Greenhouse Gas Emissions

6.3.1.1 Light-Duty Vehicles

Light-duty vehicles are defined as vehicles with a gross vehicle weight rating of 8,500 lb or less (USDOE 2012), which includes passenger cars and light trucks, such as small pickups, sport utility vehicles, and vans, most of which are used for personal purposes. Light-duty vehicles are mostly powered by spark-ignition engines fueled with gasoline. Performance of these engines can be improved through addition of octane boosters to gasoline.

Until the 1970s, the preferred octane booster was tetraethyl lead, but it was subsequently phased out because of its toxicity. While aromatics derived from petroleum, such as benzene, toluene, and xylene, can also increase octane rating in gasoline, they are relatively expensive to produce (Yang et al. 2019) and highly toxic to humans and the environment, contributing to 96% of particulate matter emissions from gasoline (Detchon and Modlin 2021). Oxygenated compounds, such as methyl tertiary butyl ether (MTBE), methanol, and ethanol, can boost octane as well as reduce volatile organic and particulate matter emissions from internal combustion engines. The petroleum-refining industry preferred MTBE produced from petroleum as oxygenated octane booster until it was phased out starting in 2000 as leaks from underground storage tanks and spills during transport were responsible for ground water pollution. The rise of the grain ethanol industry corresponded to this phase-out.

Ethanol is now blended with virtually all gasoline sold in the United States, but typically at only 10% volume (known as E10) (Brown and Brown 2012). In principle, higher blends of ethanol and gasoline have the dual advantage of reducing net carbon dioxide emissions and improving engine performance. Grain-based E10 reduces carbon dioxide emissions by only 4.3% compared to neat gasoline, reflecting the small portion of renewable fuel in the E10 blend. Emissions decrease by 36% for a vehicle fueled by an 85% blend of ethanol with gasoline (E85). Many vehicles manufactured in the US, especially light-duty trucks, are able to use E85. In Brazil, truly fuel flexible vehicles have been deployed, able to burn fuel mixtures ranging from 100% gasoline to 100% ethanol. Use of ethanol produced from cellulosic feedstocks would reduce carbon dioxide emissions from light-duty vehicles by as much as 86% (Wang et al. 2007; Argonne National Laboratory 2017).

Consumer acceptance of high ethanol blends in the United States has been limited, probably reflecting the reduced driving range compared to gasoline resulting from the fact that the volumetric energy content of ethanol is only 66% that of gasoline. However, the higher-octane rating of ethanol compared to gasoline (109 versus 91–101), together with other differences between the fuels, present an opportunity to substantially offset the lower energy content of ethanol. Higher octane fuel allows engines to be designed to operate at higher compression ratios, which results in higher engine efficiency and in driving ranges approaching 80% of those of gasoline-fueled vehicles (Brown and Brown 2014; Leone et al. 2015). Similarly, methanol has the potential to increase octane ratings in fuels. Other low-carbon alternatives include the utilization of hydrogen in the form of fuel cells and renewable natural gas. These other options can be produced using a variety of biomass and waste products, all of which have great potential to reduce GHG emissions.

6.3.1.2 Heavy-Duty Vehicles

Heavy-duty vehicles are vehicles that have a gross vehicle weight rating from 8,500 to over 60,000 lb (USDOE 2012). Heavy-duty vehicles are often commercial vehicles such as utility vans, delivery trucks, buses, tractors, and sleepers. Traditionally, most heavy-duty vehicles run on compression-ignition engines that burn diesel fuel. Prior to the promulgation of environmental regulations, diesel was the primary source of soot and fine particulate emissions in the United States (USEPA 2021a). Consequently, regulations in the United States and other parts of the world were imposed on diesel to reduce sulfur emissions to as low as 10–15 ppm, leading to the introduction of ultra-low sulfur diesel, which is the standard diesel sold today. However, it has been reported that diesel fuel consumption in 2019 alone resulted in 456 million Mt of carbon dioxide emissions, representing about 24% of the total emissions in the transportation sector (USEIA 2020b).

To reduce carbon emissions and address other air pollution concerns, alternatives are sought for petroleum-derived distillate fuel, which made up 37% of global transportation energy consumption in 2012. While battery-powered electric vehicles have made impressive strides in replacing gasoline-powered spark ignition engines for light-duty vehicle applications, batteries have intrinsic limitations in range and power that make replacement of internal combustion engines for heavy-duty transportation (trucking and the maritime industry) problematic for the foreseeable future (Smith et al. 2019). Low-carbon alternatives have been developed from lipids derived from waste fats, oils, and grease and oil crops, which resemble the long-chain hydrocarbons found in distillate fuel. These include biodiesel, produced by transesterification of lipids with methanol, and renewable diesel, produced by hydro-treating lipids with hydrogen gas.

A blend of 20% biodiesel and 80% petroleum diesel, known as B20, can reduce carbon dioxide emissions by 15%, while neat biodiesel (B100) provides an additional 59% reduction of GHG emissions (Alleman et al. 2016). Renewable diesel has the potential to reduce emissions by 65% or more, depending on feedstock (Huo et al. 2008). In the face of high demand for lipid feedstocks, thermochemical processes that convert lignocellulosic biomass, such as wood and crop residues, into “cellulosic” renewable diesel are of considerable interest (Smith et al. 2019).

Recent studies have also shown that ethanol can be burned in diesel engines through mixing-controlled compression ignition, resulting in anticipated GHG emissions saving of 45% (Blumreiter et al. 2019). Other alternatives include compressed natural gas and compressed hydrogen, which offer the potential to reduce emissions by over 100% and 86%, respectively (Mintz et al. 2010; Han et al. 2011; Lee et al. 2018).

6.3.1.3 Aviation

The aviation industry includes civil and military sectors. Civil aviation encompasses passenger aircraft and general aviation, the latter includes small agricultural aircraft such as crop dusters. Military aviation includes small surveillance aircraft to larger transport aircraft. Although adoption of battery-powered electric vehicles for light-duty vehicles is projected to grow rapidly, electrification of aviation is problematic. Jet fuel has an energy density of 5.4 kWh per lb (43 MJ/kg), which enables construction of large, long-range aircraft. This energy density is nearly 60 times that of the batteries used in today’s electric vehicles (Holladay et al. 2020).

In 2019, 26 billion gallons of jet fuel were used in the US, while 106 billion gallons were used worldwide (USEIA 2020c). The worldwide market is expected to expand to 230 billion gallons by 2050. Commercial aviation is responsible for approximately 13% of the GHG emissions from the transportation sector (Holladay et al. 2020).

Growing demand for aviation fuel and increased environmental awareness has encouraged airlines to increase fuel efficiency while decreasing emissions. Because the demand of the jet fuel market is expected to more than double over the next several decades, a significant increase in production of low-carbon jet fuels, known as sustainable aviation fuel, will be essential to reduce GHG emissions. Airlines have pledged to reduce carbon dioxide emissions by 50% by 2050 (IATA 2021).

Several kinds of sustainable aviation fuel have been developed and approved for aviation, including synthetic paraffinic kerosene, synthetic iso-paraffin, cycloalkanes, and aromatic fuels from Fischer Tropsch, alcohol-to-jet, and hydro-processing pathways of biomass or lipids (Holladay et al. 2020).

6.3.1.4 Maritime Transport

Maritime transport is the movement of goods and people over bodies of water. Specific attention is paid to shipping and cargo containers along with large passenger cruise ships. The shipping industry is responsible for 80–90% of the worldwide import and export of goods. Furthermore, international freight transport is responsible for 80% of energy consumption in maritime transport, which is expected to grow over the next two decades (Vyas et al. 2013). Heavy fuel oil, also known as bunker fuel, represents 81% of the fuel used by the maritime industry, the rest being mostly marine diesel (Vyas et al. 2013).

Maritime shipping accounts for 9.3% of carbon dioxide emissions from the transport sector and around 3% of global carbon dioxide emissions (Gielen et al. 2019). These emissions increased by 10% from 2012 to 2018 and are expected to increase as much as 50% by 2050 (EPRS 2020). Heavy fuel oil is not a particularly clean fuel, resulting in significant emissions of nitrogen oxides and sulfur oxides. Approximately, 15% of global nitrogen oxides and 13% of the global sulfur oxides emissions come from maritime shipping (Gielen et al. 2019).

In 2020, the International Maritime Organization implemented stricter limits on emissions by limiting the sulfur content in marine fuels from 3.5 wt.% of the fuel to 0.5 wt.% (USEIA 2019b). Stricter emissions are opening the door for alternative energy in the maritime transportation sector. To decrease sulfur emission, low-sulfur fuel oil and liquefied natural gas have been substituted for high-sulfur bunker fuel. However, these substitutions do not address carbon dioxide emissions from the maritime industry. The power and range requirements of large ships limit the role of electrification in maritime transport. Renewable alternatives include renewable natural gas, hydrogen, and renewable fuel oils from biomass.

6.3.1.5 Renewable Power

Maintaining the reliability of the power grid with increasing intermittent renewable power is an ongoing challenge. Greater investments in transmission, in energy storage, and in advanced distribution systems engineering to incorporate small, distributed energy resources will help to ensure high reliability.

While the focus of this chapter is on the potential for decarbonization via agriculturally based renewable fuels, the production and use of these fuels interact with the power grid that is also transitioning to renewable energy. Renewable power systems range from large wind farms, with 57% of Iowa's electricity consumption coming from wind in 2020 (USEIA 2021), to small, distributed energy resources on low-voltage distribution systems. In 2020, the Federal Energy Regulatory Commission (FERC) issued Order 2222, requiring distributed energy resources to be able to participate in all aspects of wholesale energy markets (FERC 2021). This order allows aggregators to bring portfolios of small, distributed energy resources to markets. Energy storage resources, too, can participate in energy markets on an equal footing with other resources as a result of FERC Order 841 (FERC 2018).

The significance of these two orders for Iowa's decarbonization goals is that small renewable energy production and small renewable energy storage facilities can sell into energy markets. In addition, because of variability in the marginal pricing of energy among regions, renewable fuel producers may be able to have flexible production in order to take advantage of very low energy prices that occur in Iowa during periods of high wind and low load (MISO 2021). This ability to have flexible production, storage, and use of renewable fuels (including for electricity production) will be a key component to decreasing costs and leveraging the value of these fuels while contributing to Iowa's decarbonization (Moriarty and Honnery 2016).

Dispatchable renewable power from combustion of biomass-derived fuels represent forms of energy storage. Dispatchable renewable power implies electricity generation that, on demand, can be rapidly ramped up or down (Lovegrove et al. 2018). Combustion power systems vary tremendously in their flexibility to provide power, both in terms of minimum stable output (turndown capability) and ramp rate in adjusting power output. Steam power plants capable of burning solid biomass at best have ramp rates of 4–8% per min but can be as low as 0.6% per min. In the case of a cold start of a plant, 2–7 hours are required to reach operating temperatures (Impram et al. 2020). Ramp rates for gas turbine power plants are much higher at 6–15% per min with lead times for cold start-up of only 1–2 hours. Among the quickest are diesel generators burning gaseous or liquid fuels, which can reach fuel-rated power in less than 10 seconds from a cold start. Solid biomass can be thermochemically converted to gaseous or liquid fuels via gasification or pyrolysis or biologically to gaseous fuels via anaerobic digestion (Brown and Brown 2014).

6.3.1.6 Process and Residential Heating

Heat accounts for 50% of the end-use energy consumption worldwide, which is about twice that of either electric power (20%) or transportation (30%) (IEA 2020). Heat energy can be categorized according to usage: industrial (or process) heating, residential heating, or agricultural heating. Process heat refers to the use of heat in manufacturing such as steel or cement production. The majority of process heat is produced by the combustion of fossil fuels, such as coal, oil, and natural gas, in combustion heaters and boilers. Residential heating consists mainly of space and water heating in addition to heating energy for appliances. Natural gas is the main energy source for residential heating. In 2020, industrial processes accounted for 50% of the total heat consumed while residential heating accounted for 47%. The remaining 3% of heating was used for agriculture, mainly greenhouses (IEA 2020).

The power and heat generation sector is responsible for around 40% of carbon dioxide emissions by sector as a result of combustion of fossil fuels (IEA 2019). The demand for heat and power by the residential sector is expected to continue to grow with the increase of multifamily households, predicted to grow by 25% by 2050. More efficient energy systems are required to meet this demand while limiting GHG emissions. For example, combined heat and power systems using biomass-derived fuels can achieve efficiencies over 70% while reducing carbon dioxide emissions (Huang et al. 2013; Pröll and

Zerobin 2019). Biomass offers further opportunity to expand renewable energy contribution to the growing heat demand while also decarbonizing and reducing GHG emissions. Potential renewable energy carriers for process and residential heating include hydrogen, renewable natural gas, and renewable fuels.

6.3.1.7 Chemical Manufacturing

Some chemical manufacturing industries release carbon dioxide from both the burning of fossil fuels and from the chemical transformations of carbon-bearing materials. Most prominent among these industries are the manufactures of steel (reduction of iron ore), concrete (calcination of limestone), and nitrogen fertilizer (steam reforming of natural gas). The carbon footprint of steel production can be dramatically decreased by using green hydrogen (generated from electrolysis of water using renewable energy) to reduce iron ore instead of using metallurgical coal. The GHG emissions from concrete production can be mitigated by alternatives to using calcined limestone as the cement component of concrete, which is an area of active research (Jacoby 2020). Alternatively, in some applications asphalt produced from plant-derived materials (sometimes referred to as bio-asphalt) can be substituted for concrete (Mahssin et al. 2021). In principle, renewable natural gas could be substituted for natural gas in the production of nitrogen fertilizers (Zhao et al. 2020), although hydrogen from electrolysis of water using inexpensive renewable electricity is increasingly attractive (Service 2018).

6.3.2 Low-Carbon Energy Carriers

Energy carriers are convenient and clean burning forms of energy suitable for use in the transportation, industrial, and residential sectors of the economy. They are often manufactured from “primary energy sources” that are usually inconvenient or too dirty to employ directly to provide fuel, heat, or power to society. For example, petroleum is a viscous liquid containing a wide range of hydrocarbons and contaminants that represents a primary energy source unsuitable for direct combustion in automobiles or home furnaces. However, it can be refined into gasoline, diesel, and aviation fuel with uniform properties suitable as an energy carrier. This section describes energy carriers suitable for reducing GHG emissions across the various sectors of the economy.

6.3.2.1 Hydrogen

Hydrogen can be used as energy carrier for heat, power, and transportation applications and serve as reagent in chemical synthesis of liquid fuels (such as methanol), fertilizer (ammonia), and as reducing agent in metallurgy (steel production). Hydrogen fuel cells are potentially competitive with batteries in electric vehicles for high-power, long-range applications (Fan et al. 2021). Hydrogen can also be used for energy storage, important for large-scale storage of intermittent wind and solar energy.

Hydrogen can be produced in several different ways, with distinctly different carbon footprints. Colors have been assigned to hydrogen depending on how it is produced. Traditionally, hydrogen is produced from fossil fuels such as coal and natural gas. Hydrogen produced via coal gasification is known as brown hydrogen, while hydrogen produced via reacting natural gas with steam (methane reforming) is known as gray hydrogen. Economics currently favor production of gray hydrogen, which is approximately 95% of all hydrogen produced (USDOE 2021). However, its production contributes about 830 Mt of carbon dioxide globally (Newborough and Cooley 2020).

Hydrogen produced from electrolysis of water using solar, wind, or biomass power is referred to as green hydrogen because of its low carbon dioxide emissions (about 85% lower than gray hydrogen, see

Fig. 6.2). The falling prices of solar and wind power would seem to make green hydrogen attractive, but it is still more expensive than gray hydrogen, selling for \$2.5–\$4.5 per kg H_2. Scaling of electrolysis technology is expected to decrease these costs (Lee et al. 2018). Green hydrogen could help overcome the intermittency of solar and wind power if it was stored and used to generate electricity during periods of low wind and solar power. Hydrogen from nuclear energy also has very low carbon dioxide emissions and, in principle, qualifies as green hydrogen although it is sometimes distinguished as yellow hydrogen (Baker McKenzie 2020).

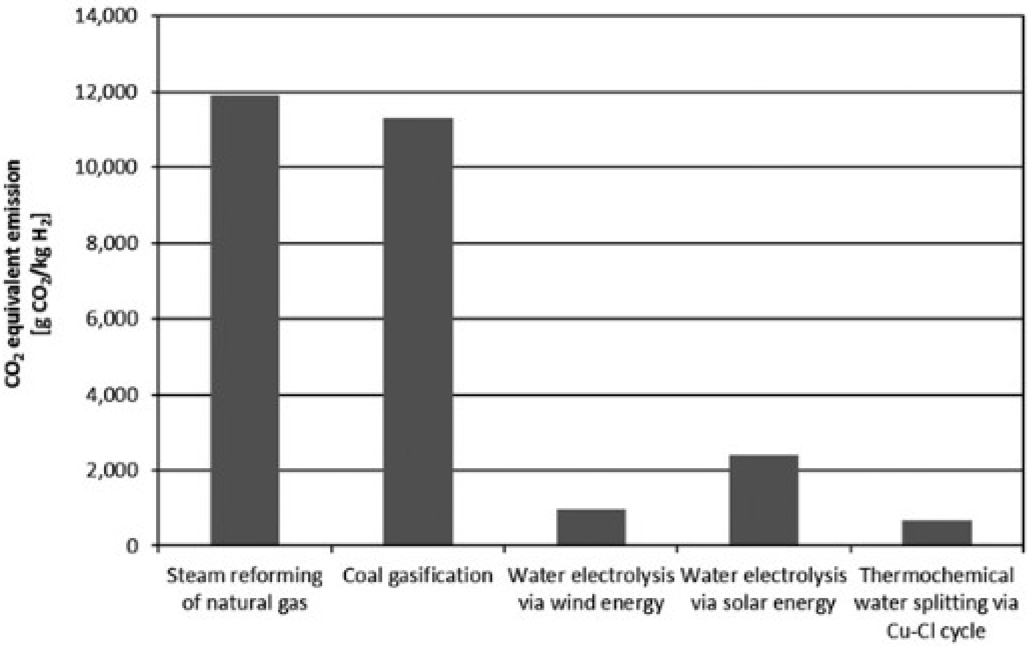

Figure 6.2. Greenhouse gas emissions from green hydrogen are significantly lower than from gray or brown hydrogen. Gray hydrogen is produced from steam reforming natural gas, while brown hydrogen is produced from coal gasification (Cetinkaya et al. 2012).

Blue hydrogen, like brown and gray hydrogen, is produced from coal and natural gas but with the important difference that the carbon dioxide emissions are captured and sequestered (Howarth and Jacobson 2021), qualifying as a GHG emissions reduction scheme for fossil fuels (Howarth and Jacobson 2021). In some instances, blue hydrogen can be produced by retrofitting existing fossil fuel power plants or processing facilities. However, the energy and cost penalties of such retrofits are significant. Adding carbon capture to the flue gas stream from pulverized coal boilers would decrease power plant thermal efficiency by 11%–23% and increase electricity costs from \$0.084 per kWh to \$0.14–\$0.16 per kWh (Supekar and Skerlos 2015). Significantly, a number of integrated coal gasification/electric power generation/carbon sequestration demonstration plants have been planned and/or abandoned in the last several years, as detailed in a 2016 MIT report (Herzog 2016). As stated by the author, “The current technologies just seem too expensive for the power sector.” However, the report also notes that many of these projects were developed during a period when gas and oil prices dramatically decreased, contributing to the unattractiveness of coal power with or without carbon sequestration.

The final color of hydrogen is turquoise, produced from electrically driven pyrolysis of natural gas along with solid carbon (Noussan et al. 2020). Turquoise hydrogen from natural gas is reported to have 49%

fewer GHG emissions than gray hydrogen (Gribova and Giese 2021). If renewable electricity is used to drive the process, turquoise hydrogen can be carbon neutral or even carbon negative if renewable natural gas is substituted for fossil-based natural gas. An important advantage of turquoise hydrogen compared to gray hydrogen is conversion of carbon into a solid material rather than gaseous carbon dioxide, which is easier to sequester than gas (Fulcheri 2021).

Hydrogen has been proposed as transportation fuel, but because of its very low density compared to liquid fuels, it would have to be used as highly compressed gas or cryogenic liquid. In principle, it could be burned in the existing internal combustion engines of both light-duty and heavy-duty vehicles. However, the low density of even compressed and liquefied hydrogen makes fuel cells more attractive than internal combustion engines for this fuel. Fuel cells are essentially batteries continuously fed with fuel instead of being periodically recharged with electricity. Electric vehicles powered with hydrogen fuel cell offer fuel economies as much as 50% higher than hydrogen internal combustion engine vehicles and thus achieve a much-improved driving range for a given volume of hydrogen fuel. Like battery-powered electric vehicles, fuel cell electric vehicles are considered "zero-emission" vehicles. While hydrogen is currently more expensive than electricity, hydrogen fuel cell-powered electric vehicles have distinct advantages over battery-powered electric vehicles, including greater range and faster refueling. In 3–5 minutes, liquid hydrogen refueling stations can transfer 5 to 7 kg of hydrogen needed for 300 miles of driving range vs around 6–8 hours using a Level-2 charger or 30 minutes for a Level-3 charger to achieve 100 miles of driving range for a typical Nissan Leaf battery-powered electric vehicle (Yilmaz and Krein 2013). Argonne National Laboratory conducted a study on the "well-to-wheel" GHG emissions of fuel cell-powered medium trucks fueled by hydrogen. They found that green hydrogen produced via solar-powered electrolysis reduced fossil fuel consumption and GHG emissions by 90% and 86%, respectively, relative to gray hydrogen (Lee et al. 2018).

Hydrogen has also been proposed as fuel for maritime transport (Gielen et al. 2019). Hydrogen would help meet maritime reduction targets for carbon dioxide, sulfur oxides, and nitrogen oxides emissions. Challenges with hydrogen as fuel in large shipping vessels are both bunkering infrastructure and storage (Van Hoecke et al. 2021) because of its low density compared to liquid hydrocarbon fuels. Methods for hydrogen storage include fuel cells, compressed hydrogen, liquid hydrogen, solid-state hydrogen carriers, and conversion to other energy carriers such as ammonia, renewable natural gas, methanol, and Fischer-Tropsch fuels. The clean footprint of hydrogen drives interest in the maritime industry, but the current lack of infrastructure has led to use of alternative energy carriers, as discussed in **Section. 6.3.1.4** on maritime transport.

One of the most compelling applications of hydrogen is energy storage for the electric utility industry. The intermittency of solar and electric power must be balanced by means to store electrical energy when its supply exceeds demand. In principle, the same battery technology used in battery-powered electric vehicles could be used for utility-scale storage of electricity. In practice, the energy density of batteries falls short of what is required to meet power interruptions longer than a few hours. In contrast, in a system called "power-to-gas," hydrogen produced from electrolysis and stored as compressed gas could fuel combustion gas turbines (McMillan et al. 2016; Kåberger 2018) or fuel cells to provide continuous back-up power for days and even weeks (Moore and Shabani 2016) as shown in Figure 6.3.

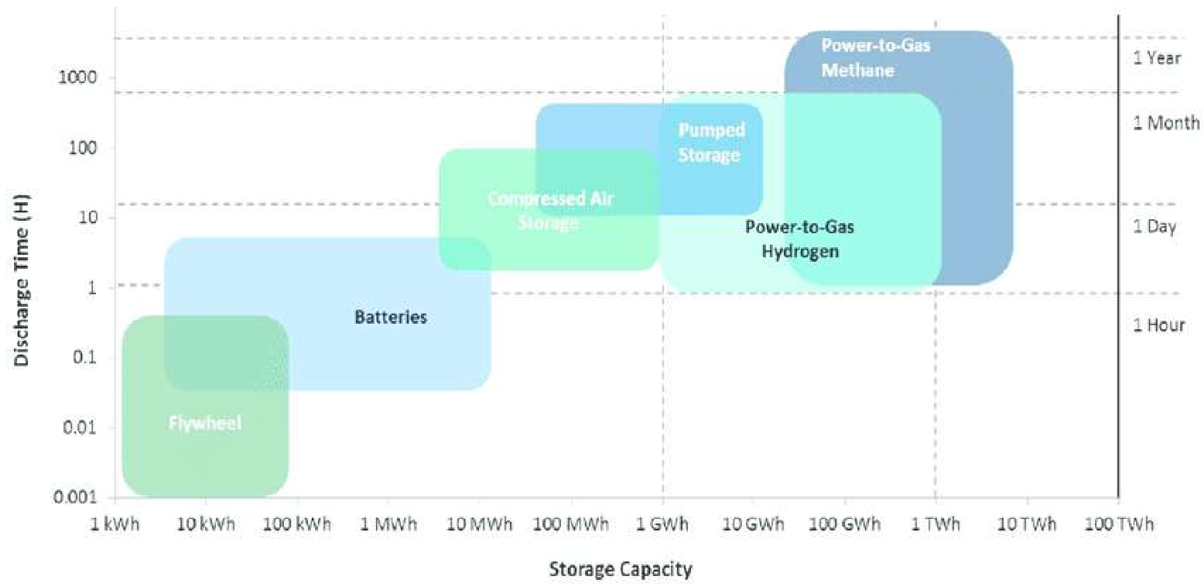

Figure 6.3. Discharge time relative to energy storage capacity for different applications (Moore and Shabani 2016).

6.3.2.2 Renewable Natural Gas

Renewable natural gas (RNG), also known as biomethane, is produced by refining biogas from the anaerobic digestion of organic materials into pipeline-quality methane. RNG is chemically identical to fossil natural gas and therefore compatible with existing natural gas infrastructure (pipelines, storage tanks, dispensing equipment, internal combustion engines, process heaters, furnaces, stoves, etc.). Despite these several potential applications, RNG is most frequently employed in the transportation sector where energy prices are higher than in other energy sectors (Parker et al. 2017). RNG also qualifies as cellulosic biofuel under the United States Renewable Fuel Standard and achieves a very low carbon intensity under California's Low Carbon Fuel Standard (Scheitrum et al. 2017).

The two main pathways to RNG are capturing and upgrading landfill gas and conversion of organic materials, such as livestock manure, municipal solid waste, and agricultural residues, in anaerobic digesters. Landfill gas is generated when municipal solid wastes decompose anaerobically. Traditionally, this methane-rich gas is flared, keeping this powerful GHG from being emitted into the atmosphere. By upgrading landfill gas into pipeline-quality methane, its energy value can be recovered while avoiding GHG emissions. With careful monitoring of operating conditions, anaerobic digesters can decompose a wide range of organic materials, including livestock manure, municipal wastewater sludge, food wastes, industrial wastewater, and fats, oils and grease (American Biogas Council 2021). Biogas from anaerobic digestion is 50–70 vol% methane and 30–40 vol% carbon dioxide with trace amounts of hydrogen sulfide and other gases (USEPA 2021b). Anaerobic digestion also produces a solid residue known as digestate as a co-product. Digestate is rich in nutrients and can be used as fertilizer, livestock bedding, or soil amendment (USEPA 2021c). To upgrade biogas from landfills or anaerobic digestors into RNG, methane is separated from the other gases and compressed to pressures suitable for injection into natural gas pipelines (Han et al. 2011; USEPA 2020).

RNG can be directly substituted for fossil-based natural gas in the engines of light-duty or heavy-duty vehicles modified to operate on compressed or liquefied natural gas. With an octane rating of 130, comparable to methanol and ethanol, RNG can be advantageously employed in high-compression engines to achieve increased engine efficiently. An incentive for consumers is its low cost compared to gasoline. Natural gas is often 11–53% cheaper than gasoline and diesel (Werpy et al. 2010; Whyatt 2010). The fact that many homes are connected to natural gas pipelines suggests that home refueling of natural gas vehicles is a possibility. However, the prices paid for residential natural gas tend to be higher than commercial prices, and expensive pressure boosters would have to be added because the delivery pressure of natural gas to homes is too low for vehicle applications.

There are three types of fuel systems that can be used with RNG: dedicated engine (vehicle runs only on natural gas); bi-fuel (vehicle runs on both gasoline and natural gas); and dual-fuel (vehicle requires diesel to assist with ignition but otherwise is fueled by natural gas). The first two options are available for light-duty vehicles, while the last option is more suited for heavy-duty vehicles. The use of natural gas in compression-ignition engines of heavy-duty vehicles is motivated by the high emissions of particulate matter and nitrogen oxides from the diesel fuels used in their diesel engines. Because natural gas does not auto-ignite in a compression-ignition engine, a small amount of diesel fuel is used to start the ignition, after which the vehicle runs fully on natural gas. This system is able to substitute RNG for 90% of the diesel fuel normally burned in compression-ignition engines (Chala et al. 2018).

On average, replacing diesel fuel with fossil-based natural gas can reduce well-to-wheel GHG emissions by 14%. When RNG is produced from land-fill gas, GHG reductions average 71%. For RNG generated from anaerobic digestion of livestock manure, a reduction of 100–108% is achieved. Use of natural gas in light-duty vehicles results in well-to-wheel GHG emissions of only 11% of gasoline. RNG produced from landfill gas reduces GHG emissions by 82% (Mintz et al. 2010; Han et al. 2011).

6.3.2.3 Green Ammonia

Ammonia (NH_3) is one of the most important inorganic materials used in human societies because of its role as nitrogen fertilizer. In 2018, ammonia production worldwide reached 140 million Mt, with the United States accounting for approximately 9% of total production (Zhang et al. 2020). Eighty percent is used for making fertilizer, with the remainder being used in production of fibers, refrigerants, and sorbents (Patonia and Poudineh 2020; Faria 2021). While ammonia itself does not contain carbon, conventional ammonia production is not a green process as approximately two thirds of ammonia is currently synthesized from hydrogen derived from natural gas (Zhang et al. 2020). Most of the rest is synthesized from hydrogen derived from coal, fuel oil, or other hydrocarbons. Ammonia synthesized from green hydrogen (derived from renewable resources) would dramatically reduce GHG emissions and is thus referred to as green ammonia.

Ammonia is produced by reacting nitrogen and hydrogen at high pressure over a catalyst via the Haber-Bosch process (Salmon and Bañares-Alcántara 2021). Green hydrogen for synthesis of green ammonia can be produced through either biomass gasification or water electrolysis with electricity from renewable power. In a process analogous to the production of hydrogen from fossil fuels, gasification of biomass produces a mixture of hydrogen and carbon monoxide that can be further upgraded and purified to hydrogen. While the efficiency of green ammonia production from biomass largely depends on energy integration within the gasification plant, it is estimated to be approximately 50% (Zhang et al. 2020). Because the co-product carbon dioxide is derived from biogenic carbon, it would make the hydrogen carbon negative. Green hydrogen from electrolysis of water using renewable electrical power only achieves carbon neutrality, but has the advantage of higher overall energy efficiency in the production of green ammonia, approaching 75% (Zhang et al. 2020). The nitrogen required for synthesis of ammonia via the Haber-Bosch process is produced by cryogenic air separation or pressure swing adsorption. The substantial energy requirement to recover nitrogen from air should be provided from renewable energy to minimize net GHG emissions (Patonia and Poudineh 2020).

Green ammonia would significantly reduce the carbon footprint of agriculture compared to the use of conventional ammonia as nitrogen fertilizer. However, several properties of green ammonia make it competitive with green hydrogen as a low-carbon energy carrier. As shown in Table 6.2, compared to compressed hydrogen, ammonia's volumetric energy density is 2.4 times higher; it can be stored at lower pressure; and its explosion limits are much narrower and it is thus less dangerous (Aziz et al. 2020). Green ammonia could be used as fuel for electric power generation, solid oxide fuel cells, internal combustion engines, and gas turbines (Morlanés et al. 2021).

Table 6.2. Comparison of compressed hydrogen and liquid ammonia characteristics (Aziz et al. 2020).

Properties	Unit	Compressed Hydrogen	Liquid Ammonia
Storage Method	-	Compression	Liquefaction
Temperature	°C	25 (room)	25 (room)
Storage Pressure	MPa	69	0.99
Density	kg/m^3	39	600
Explosive Limit in Air	%vol	4-75	15-28
Gravimetric Energy Density	MJ/kg	120	18.6
Volumetric Energy Density	MJ/m^3	4680	11,160

The cost of producing green ammonia is currently \$400–\$450 per ton with projections of \$300 per ton by 2040 (Zhang et al. 2020; Cesaro et al. 2021). While the production cost of green ammonia is currently higher than that of brown or gray ammonia, the push for decarbonization may improve its prospects (Chehade and Dincer 2021). Besides the high cost, green ammonia is commonly produced via the Haber Bosch process which loses some energy as heat during the synthesis of ammonia, resulting in the overall energy efficiency for producing ammonia being lower than for producing compressed hydrogen.

While pure ammonia's high autoignition temperature, low flame speed, and emissions of nitrous oxides present challenges in combustion applications, it can be blended with other fuels to mitigate these effects. A mixture of green ammonia and green hydrogen substantially reduces emissions from several combustion devices, ranging from spark ignition engines to industrial gas turbines (Chehade and Dincer 2021). For example, a mixture of 70% ammonia and 30% hydrogen burned in a gas turbine showed optimal stability and performance (Cesaro et al. 2021). Green ammonia is considered a viable, low-carbon fuel for maritime transport because of its relatively high density and relatively simple storage requirements compared to either compressed or liquefied hydrogen. Some forecasts suggest ammonia could provide 25–99% of global shipping fuel by 2050 (Ash and Scarbrough 2019; Cesaro et al. 2021).

6.3.2.4 High-Octane, Low-Carbon Fuels

Despite advances in battery-powered electric vehicles, it will take several decades to replace the current vehicle fleet. Thus, low carbon, liquid fuel substitutes for gasoline and diesel will play important roles in decarbonizing the economy (Debnath et al. 2019). However, these low-carbon alternatives must also achieve high fuel economy to keep up with increasingly stringent Corporate Average Fuel Efficiency (CAFE) standards. High-octane, low-carbon fuels (HOLCFs) can achieve both low carbon emissions and high fuel economies. HOLCFs are conventional hydrocarbon fuels, such as gasoline or diesel, blended with 20–50% oxygenated fuels, such as ethanol, produced from renewable sources. Examples of mature renewable oxygenates include ethanol and methanol while alternative oxygenates include propanol, butanol, and furans (Ershov et al. 2021). Although these alternative oxygenates have some attractive features in fuel blends, their relatively low yields in production from biorenewables make them unattractive at present (Yanowitz et al. 2011). The rest of this section focuses on ethanol and methanol as high-octane fuels.

Ethanol is the most common form of oxygenate, produced via fermentation from corn, sugarcane, and other kinds of sugar and starch crops. In the United States, ethanol is almost universally blended with gasoline. Blending higher levels of ethanol into gasoline is advantageous as it has the ability to increase octane ratings of fuel, boosting engine efficiency while reducing GHG emissions (Johnson et al. 2015). Ethanol has an average octane rating of 109, while gasoline has an octane rating of only 91 to 93 (Anderson et al. 2012). While most ethanol is currently produced from corn starch, producing ethanol from agricultural residues, such as corn stover, or energy crops such as switchgrass and Miscanthus, has more potential to decrease GHG emissions. Studies have reported that ethanol from corn stover can reduce GHG emissions by 80–100%, while ethanol derived from switchgrass, Miscanthus, and woody biomass has the potential to reduce emissions by 50–115% (Wang et al. 2012; Morlanés et al. 2021).

A typical HOLCF is a mid-level ethanol-blended fuel with approximately 20–40% ethanol. Researchers at United States DOE national laboratories have found that ethanol-blended HOLCFs can increase octane rating to over 100 (Note: E10 has an octane rating of 87) (Johnson et al. 2015). Increasing octane rating allows engine operation at increased compression ratios, achieving higher thermal efficiency and torque, resulting in improved automotive performance (Johnson et al. 2015). A study by Oak Ridge National Laboratory showed that gasoline blended with 30% ethanol improves fuel economy by 5.7% (Theiss et al. 2016). Figure 6.4 illustrates the GHG emission benefits from HOLCFs. Increases in fuel economy in terms of vehicle miles-per-gallon of gasoline-equivalent of 5% and 10% from use of HOLCFs can decrease GHG emissions 4% and 8% relative to conventional E10. The reported benefits of utilizing E25 and E40 blends of corn ethanol and gasoline can reduce GHG emissions by 8% and 13%, respectively. When ethanol is produced from cellulosic ethanol, additional GHG emissions reductions of 3%, 12%, and 23% for E10, E25, and E40, respectively, are possible (Theiss et al. 2016).

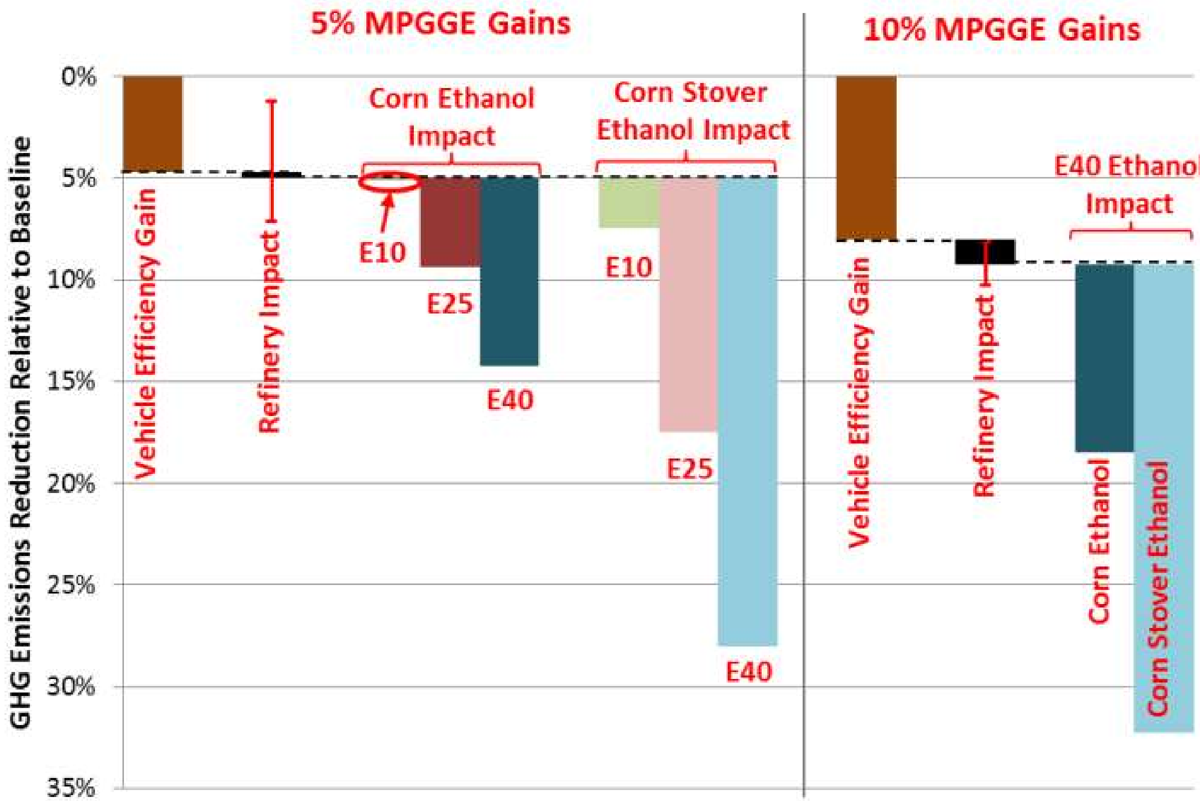

Figure 6.4. Well-to-wheel greenhouse gas (GHG) emissions reductions from HOLCFs for different blending levels relative to regular gasoline (E10) (Theiss et al. 2016). MPGGE = miles per gallon of gasoline-equivalent.

Although questions are sometimes raised about ethanol's compatibility with fuel systems designed for gasoline, flex-fuel vehicles can operate on ethanol blends as high as 85%. Therefore, mid-level blends of 20–40% can be used in all 18 million flex-fuel vehicles in the country. HOFC market penetration will likely be constrained by non-technical barriers such as cost, station knowledge on refueling equipment, and permitting of new storage tanks (Theiss et al. 2016).

Methanol is an even higher-octane fuel than ethanol, with octane ratings of 129 to 134 (Harris et al. 2021). However, most methanol is currently produced from reforming natural gas or gasification of coal, resulting in much higher GHG emissions than ethanol (Rivarolo et al. 2016). In principle, it could be produced from renewable resources including forestry and agricultural residues, municipal wastes, black liquor, renewable natural gas, and synthesis from green hydrogen and captured carbon dioxide (Zhen and Wang 2015; Svanberg et al. 2018; Roode-Gutzmer et al. 2019).

China is a leader in developing methanol for blending with gasoline or use as neat fuel. In contrast, the use of methanol as oxygenate in gasoline in the United States peaked in 1997, displaced by ethanol with very few flex-fuel vehicles currently employing methanol. However, the higher-octane rating and vaporization rates of methanol compared to ethanol result in higher engine efficiency and power (Mcrae and Ruppel 2011).

The GHG emission advantages of methanol compared to gasoline depend on the renewable feedstock employed in its production, as illustrated in Figure 6.5 (Hobson and Márquez 2018). Blends of 85 vol%

methanol and 15 vol% gasoline (M85) produced from landfill gas, manure biogas, and biomass gasification reduce GHG emissions by 60%, 110%, and 68%, respectively (Wang and Lee 2017). The methanol produced by Methanex Corporation in Canada produces methanol with emissions that are 30% lower than from gasoline (Hobson and Márquez 2018). Methanex's methanol cost ranged between \$0.20–0.51 per kg between 2010 and 2020. The average cost of methanol in the United States between 2018 and 2020 was approximately \$0.41 per kg. According to a recent techno-economic analysis, the cost of methanol from biomass is \$0.39 per kg, which is below the average cost of methanol in the US, suggesting a potential economically attractive pathway (Harris et al. 2021).

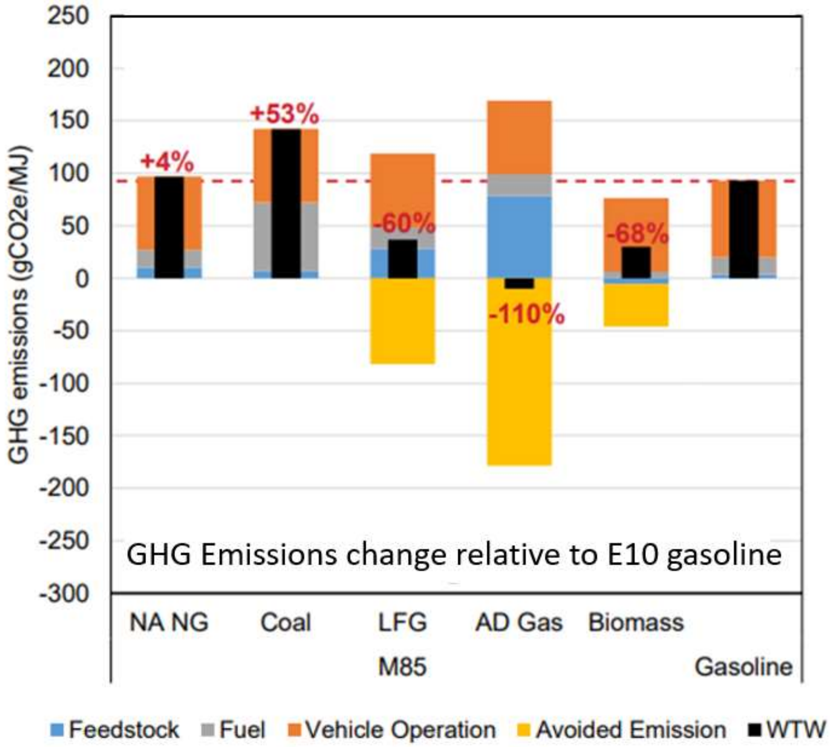

Figure 6.5. Greenhouse gas (GHG) emissions reductions relative to E10 for M85 produced from various feedstocks. Negative GHG emissions arise from processes that replace methane emissions with CO_2 emissions or remove carbon dioxide from the atmosphere (adapted from Wang and Lee, 2017). NA NG = North American natural gas; LFG = landfill gas; AD Gas = biogas from anaerobic digestion of manure; Biomass = biomass gasification; WTW = wells to wheel.

Low-level blends of methanol are compatible with most existing fuel systems. However, fuel system modifications are required for higher methanol blends because of their corrosivity to many metals and plastics (Yanowitz et al. 2011). Furthermore, methanol has half the volumetric energy density of gasoline, resulting in reduced vehicle ranges unless employed in engines optimized for the high-octane rating of methanol (Bromberg and Cheng 2010).

6.3.2.5 Ethanol as Fuel for Diesel Engines

Ethanol is most frequently employed in the spark ignition engines of light-duty vehicles to take advantage of its high-octane rating. The increasing prospect of battery-powered electric vehicles dominating future light-duty vehicles markets has raised questions about the future viability of ethanol as transportation fuel. However, research has shown that ethanol can be used in the compression ignition engines of heavy-duty vehicles, raising the possibility of ethanol as a low-carbon diesel fuel substitute (Vergel et al. 2020). Ethanol can be used in diesel engines in any of three ways: use of dual-fuel systems, fumigation, and blending with diesel. The first two options require engine modification and are more expensive. Therefore, the blending option is most attractive for near-term applications.

Ethanol blended with diesel is known as E-diesel. Up to 15% of ethanol blended in diesel is a common blend (McCormick 2002). E-diesel fuels have the potential to reduce GHG and other emissions and improve engine cold flow properties and lubricity (McCormick and Parish 2001). E-diesel can be directly mixed to desired blends, requiring no special equipment. Up to 30 vol% ethanol can be blended with diesel fuel without engine modification (Vergel et al. 2020). However, E-diesel blends have lower flash points and higher evaporation rates from fuel tanks (McCormick 2002; Vergel et al. 2020), requiring them to be handled more like gasoline than diesel. This may require modification of storage and fuel handling systems. There is also a possibility that ethanol may separate from diesel at low temperatures (McCormick 2002).

E-diesel fuels are still under development. A partnership of ClearFlame Engine Technologies and Alto Ingredients has recently demonstrated high-temperature, mixing-controlled compression ignition of ethanol in diesel engines. This technology will facilitate the adoption of high-octane, low-carbon fuels such as ethanol and methanol into diesel engines. ClearFlame Engine Technologies anticipates 45% reduction in GHG emissions from the use of ethanol in a retrofitted Class 8 truck diesel engine (Blumreiter et al. 2019).

6.3.2.6 Biodiesel

Biodiesel is produced by the reaction of lipids with methanol (a process known as transesterification) (Alleman et al. 2016). A wide range of lipid feedstocks can be employed, including canola oil, soybean oil, corn oil, recycled cooking oil, animal fats, or a combination of these feedstocks. Biodiesel is commonly blended with petroleum-derived diesel fuel at a ratio of 5 vol% (B5), which provides the lubricity needed in modern low-sulfur diesel fuels (Hazrat et al. 2015). Increasing the proportion of biodiesel in fuel blends improves engine performance and reduces GHG emissions (Alleman et al. 2016). Blends up to 20 vol% (B20) can often be employed in standard diesel engines, but higher blends may require engine modifications, and engines may experience cold-start problems compared to diesel. The GHG benefits of biodiesel are significant, with B20 and B100 achieving GHG reductions of 15% and 74%, respectively. Biodiesel has other environmental benefits, including reduced emissions of particulate matter, carbon monoxide, and unburned hydrocarbons (Moriarty 2011).

6.3.2.7 Renewable Diesel

Renewable diesel, like biodiesel, is produced from lipid feedstocks. Although more expensive to produce, it has the advantage of being fully compatible with petroleum-based diesel. Renewable diesel is produced through hydro-treatment rather than transesterification of lipid feedstocks, producing a hydrocarbon fuel that is physically and chemically indistinguishable from petroleum-derived diesel (USEIA 2020b). Because it is identical to petroleum diesel, renewable diesel is classified as a “drop-in” fuel, allowing its use with existing diesel engines and fuel infrastructure (pipelines, storage tanks, etc.). Furthermore, renewable diesel has a cloud point similar to petroleum-derived diesel, making it more suitable than biodiesel for cold climates (Knothe 2010). Renewable diesel has a cetane number similar to conventional diesel and the additional advantages of lower sulfur and aromatic content (Yoon 2011). In terms of GHG benefits, Argonne National Laboratory reported that diesel engines burning either 100%

biodiesel or renewable diesel from soybeans reduce well-to-wheel GHG emissions by as much as 66–68% and 74%, respectively (Huo et al. 2008).

The supply of waste fats and oils is much smaller than fuel demand while vegetable oils from purposely grown oil seed crops is relatively expensive. Eventually, alternative biomass resources suitable for production of renewable diesel will be required. Most relevant to the state of Iowa are crop residues consisting of lignocellulosic biomass, although this also includes woody biomass. Pyrolysis of lignocellulosic biomass produces a liquid suitable for hydro-treatment to renewable gasoline and diesel (Elliott et al. 2015). Economic analysis indicates this could be a cost-effective replacement for petroleum-based transportation fuels (Zhang et al. 2013; Brown 2017). If biochar produced during pyrolysis is sequestered, this pathway to renewable diesel is expected to be carbon negative (see **Section 6.4** on carbon removal strategies).

6.3.2.8 Sustainable Aviation Fuel

Jet fuel accounts for approximately 25% of transportation fuel consumption in the United States (Holladay et al. 2020). Electrification is not the solution for decarbonizing the aviation industry considering the range and power requirements for aviation energy carriers. Thus, unlike the light-duty vehicle sector, the only realistic option for low-carbon energy carriers for aircraft is sustainable aviation fuel (SAF). Sustainable aviation fuel is expected to be a drop-in fuel that meets all specifications for current jet fuel. Currently, there are five approved pathways for SAF that meet ASTM standard D7566 for synthetic jet fuel (Shahabuddin et al. 2020). As detailed in Figure 6.6, these include:

- HEFA-SPK – Hydrogenated esters and fatty acid fuels from cooking oils, animal fats etc.
- FT-SPK – Fischer-Tropsch fuels from biomass resources
- FT-SKA – Fischer-Tropsch fuels with aromatics from biomass resources
- SIP-SPK – Synthetic iso-paraffin from fermented hydro-processed sugar
- ATJ-SPK – Alcohol-to-jet fuels from isobutanol

Sustainable aviation fuels most relevant to resources readily available in Iowa include hydro-processed esters and fatty acids (HEFA), Fischer-Tropsch (FT) fuels, and alcohol-to-jet (ATJ). HEFA fuels are produced by hydrodeoxygenation of vegetable oils and animal fats. They are particularly promising as they can be used without blending with traditional fuels (Hari et al. 2015). Several companies, including Nest Oil Company and UOP Honeywell, are currently producing HEFA fuel for aviation. Used cooking oil is the main resource for production of HEFA fuel, setting a severe constraint on supply (Doliente et al. 2020). Like renewable diesel, which is produced from lipids in a closely related process, expansion of HEFA fuels will require hydro-processing of bio-oils from the pyrolysis of lignocellulosic biomass. This will have both economic and environmental advantages (Brown 2021).

Sustainable aviation fuels via the Fischer-Tropsch pathway involves the gasification of a wide range of feedstocks, including municipal solid waste and wood and crop residues, into a gas mixture referred to as syngas followed by Fischer-Tropsch catalytic synthesis to liquid hydrocarbons. The ability of gasification to convert a wide range of less-than-pristine feedstocks into syngas makes this pathway potentially very attractive. Furthermore, the Fischer-Tropsch process is already commercially established for converting fossil fuels into liquid hydrocarbons, although these need further upgrading to meet specifications for sustainable aviation fuel (Shahabuddin et al. 2020). Unfortunately, Fischer-

Tropsch is not cost-effective with fossil fuel feedstocks even when using inexpensive coal (Hari et al. 2015; Shahabuddin et al. 2020) and is even less attractive when processing biomass feedstocks (Swanson et al. 2010). However, Fischer-Tropsch has the potential to produce SAF with up to 95% reduction in GHG emissions (Doliente et al. 2020).

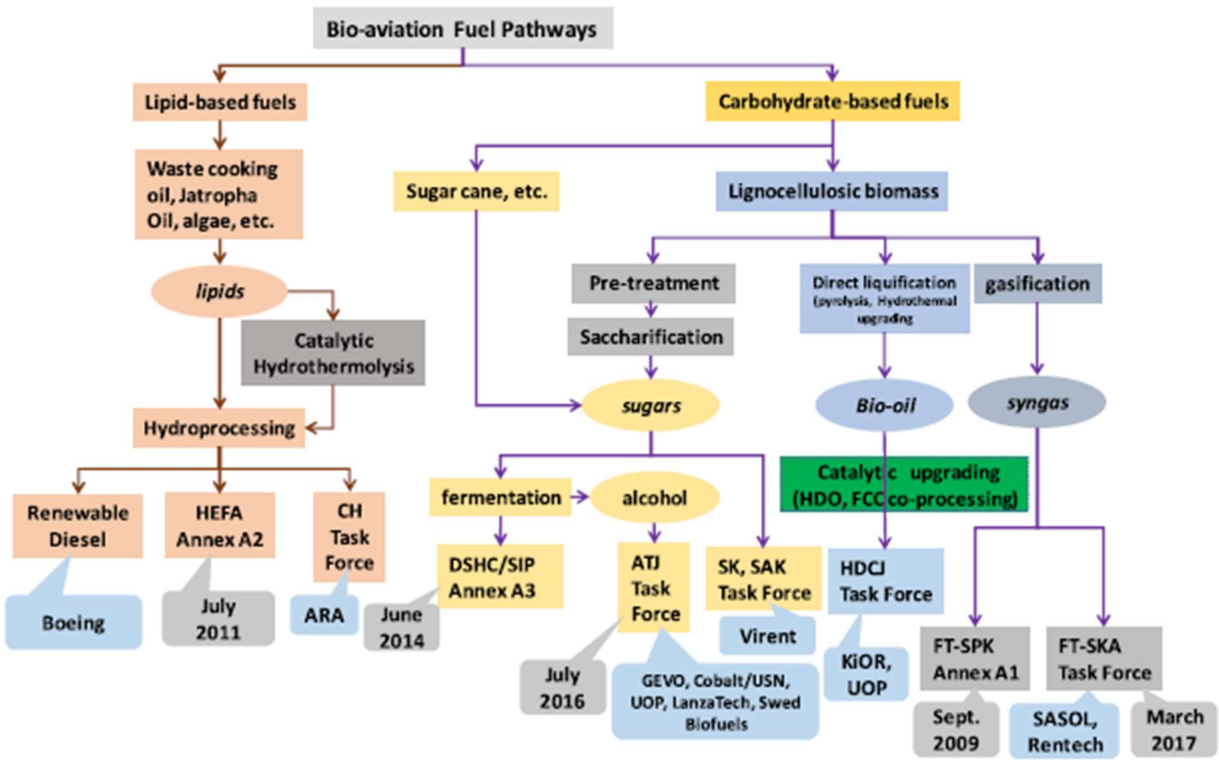

Figure 6.6. Sustainable aviation fuel pathways approved by ASTM D7566 (Wang et al. 2019).

The alcohol-to-jet pathway ferments sugars to alcohols, such as ethanol or isobutanol, followed by dehydration, oligomerization, and hydrogenation to produce sustainable aviation fuel. Although sugar and starch crops can be used, lignocellulosic biomass is attractive as a source of sugars to expand supply and improve environmental performance, including reduced GHG emissions (Holladay et al. 2020). Efforts to date to commercially produce cellulosic sugars via acid or enzymatic hydrolysis have been stymied by technical and economic challenges (Brown and Brown 2013). An alternative pathway to sugars is thermal depolymerization of biomass via pyrolysis (Brown 2021). This produces mostly anhydro sugars that can be either hydrolyzed to glucose before fermentation, or suitable microorganisms can be used to directly ferment the anhydro sugars to ethanol (Bacik and Jarboe 2016). Ethanol for the ATJ process can be produced with substrates other than sugars. For example, gas fermentation uses microorganisms to convert gas mixtures rich in carbon monoxide, hydrogen, and carbon dioxide into ethanol, which companies such as LanzaTech have successfully commercialized (Shen et al. 2015). A limitation of the alcohol-to-jet pathway is competition for ethanol in production of previously discussed HOLCFs, as discussed in **Section 6.3.2.4**.

Sustainable aviation fuel provides the only practical energy carrier for aviation. While SAF is restricted to blending ratios of 50% for HEFA and Fischer-Tropsch fuels and 30% for ATJ fuels, in principle it could completely substitute for petroleum-based jet fuel although their cost makes this difficult to justify at present. HEFA is currently the most economical SAF, costing about $1/L vs $1.80/L for ATJ from biomass (Diakakis 2019). However, availability of suitable feedstocks may be the largest factor in locating SAF production facilities, presenting an attractive opportunity for Iowa to produce SAF for distribution to Midwestern airports.

6.3.2.9 Renewable Fuel Oil

Renewable fuel oil is defined in this report as a renewable substitute for heavy fuel oil suitable for burning in furnaces or boilers to generate heat, firing in large diesel engines for stationary power, or use as bunker fuel substitute in maritime transportation. While cars, trucks, off-road vehicles, and aircraft typically require highly refined liquid fuels with relative low viscosity and flash point, large diesel engines are often designed to use inexpensive, low-quality fuels. The trade-off for using these inexpensive petroleum-derived fuels is higher exhaust emissions. For example, marine fuels account for 4–9% of sulfur oxides and 14–31% nitrous oxides emitted globally (Cortez et al. 2021). International standards are increasingly placing stringent limits on sulfur and nitrogen emissions from marine fuels. There are also calls to reduce GHG emissions from the marine industry, which is responsible for 3% of global GHG emissions.

Several pathways are under development to provide renewable marine fuels (Kass et al. 2018). Among these are refining of fats and oils (Bruun et al. 2019); gasification of biomass to syngas with upgrading to liquid hydrocarbons (Tan and Tao 2019); pyrolysis of biomass to bio-oil; solvent liquefaction of lignin (Zhang et al. 2021); and algal marine fuel (Zaky 2021). As previously described, some of these pathways produce highly refined liquid fuels more suitable as replacements for gasoline, diesel, and jet fuel and are likely too expensive to serve as widespread replacement of marine fuels.

The relatively low prices of bio-oil from fast pyrolysis of biomass or biocrude from solvent liquefaction of lignin makes them more attractive than more highly refined fuels for production of renewable fuel oil (Bridgwater 2018a; Bridgwater 2018b). However, these products show poor stability during storage as a result of their high chemical reactivity even at room temperature (Diebold 2000; Oasmaa et al. 2016). Chemical treatments that stabilize bio-oil and biocrude can improve their prospects as renewable fuel oil (Rover et al. 2015).

Figure 6.7 shows the potential reduction of sulfur oxides and GHG emissions of renewable marine fuels compared to traditional heavy fuel oil. Notably, 100% replacement of residual fuel oil from petroleum with biodiesel, renewable diesel, and Fischer-Tropsch diesel reduced GHG emissions from about 95 g carbon dioxide per MJ to 30, 25, and 5 g carbon dioxide per MJ (12 oz per kW*hr to 3.8, 3.2, and 0.64 oz carbon dioxide per kW*hr), respectively.

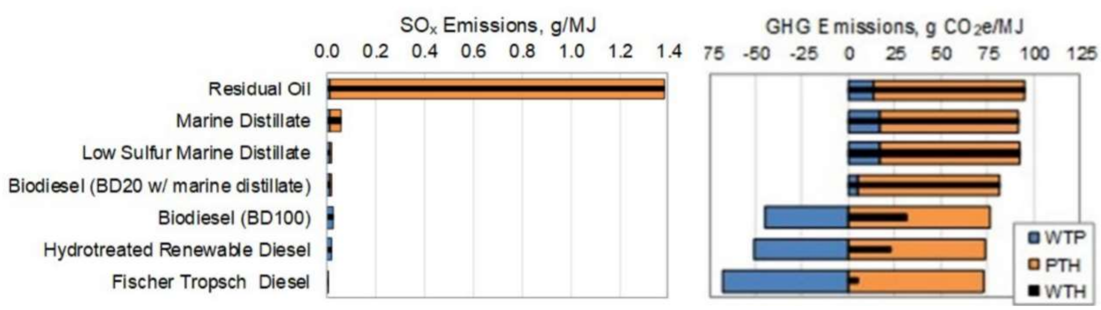

Figure 6.7. Sulfur oxides (SOx) and greenhouse gas (GHG) emissions for renewable marine fuels with well-to-pump (WTP), pump-to-hull (PTH) and well-to-hull (WTH) (Kass et al. 2018).

6.3.2.10 Nuclear Biofuels

Nuclear biofuels are produced by processing biomass into biofuels using hydrogen and heat derived from nuclear energy (Forsberg et al. 2021). Some of the chemical energy in biomass is consumed in converting it into biofuels, reducing the yield of fuel. As conventionally produced, almost half of the energy content of biomass is lost in the biomass-to-fuel conversion process. Biofuel yield could be significantly increased if the energy for processing biomass was obtained from sources other than the biomass. Solar and wind energy could potentially fill this role, but their intermittency results in low plant availability (capacity factor) and poor economic performance. The nuclear biofuel concept integrates a biorefinery (as a source of carbon to synthesize fuel) and a nuclear power plant to provide a continuous source of zero-carbon thermal energy and electricity (Plant et al. 2021). The electricity is used to produce green hydrogen required for deoxygenating biomass molecules to hydrocarbons. The resulting fuels are indistinguishable from gasoline, diesel, and aviation fuels currently produced from petroleum, making them drop-in fuels compatible with existing transportation infrastructure. Economic analysis suggests the process would only be economical for large, centralized processing plants. Given that biomass resources are highly distributed and of relatively low density, they would have to be densified for cost-effective transport to the processing plant. Pellets, bio-oil, and RNG from pelletization, fast pyrolysis, and anaerobic digestion of biomass are considered suitable densified feedstock for long-distance transport. Gasification of pellets or bio-oil or steam reforming of RNG to syngas, using nuclear thermal energy to heat the processes, followed by Fischer-Tropsch synthesis to liquid hydrocarbons, is one approach to nuclear biofuels although others have been proposed. The concept of collocating a nuclear power plant to provide low-pressure steam to a manufacturing facility has already been commercially deployed in Canada, Switzerland, and Russia (Forsberg 2009). Nuclear biofuels could be deployed within 20 years based on existing technologies.

6.4 Carbon Removal Engineering Technologies

Agricultural management practices and engineering technologies for carbon removal have been studied worldwide, but their applicability often depends on geographical location. Here we summarize the current understanding of potential **engineering technologies** for application in Iowa. Opportunities for carbon removal through **agricultural management practices** are described in **Chapter 4** of this report. We considered the engineering technologies listed in Table 6.3 applicable to the state of Iowa because of its existing infrastructure and/or agricultural capability. The carbon removal and sequestration pathways considered here exploit existing biogenic carbon dioxide waste streams or biomass currently grown that could be promoted as part of Iowa's agricultural economy. Detailed descriptions of the relevant technologies are provided in the following sections.

Table 6.3. Carbon capture engineering technologies and their potential for use in Iowa.

Technology	Near-Term Potential for Iowa
Direct Air Capture	Low
Methane Capture and Removal	Low

Bioenergy with Carbon Capture and Storage	Moderate
Capture and Sequestration of Biogenic Carbon Dioxide	High
Pyrolysis to Biochar	Moderate to High
Microbial Carbonate Precipitation in Soils	Moderate
Microbial Carbonate Precipitation in Anaerobic Digestors	Moderate
Carbon Removal in Concrete Production	Moderate

6.4.1 Direct Air Capture

Direct air capture involves passing air over an appropriate human-made sorbent material to directly capture carbon from the air (Fig. 6.8; Sanz-Perez et al. 2016; Keith et al. 2018). The sorbent is subsequently regenerated to release the carbon dioxide as a pure gas stream that is ready for transport and storage. One of the prominent advantages of direct air capture systems compared to other carbon removal strategies is that they can be geographically situated wherever there is available energy to power them (NASEM 2019). It has been suggested that they can be manufactured as small modules, capturing "economies of number" through mass production, and shipped to installation sites (Lackner 2010).

Although the technology is already used for carbon scrubbing from crew quarters in submarines and spacecraft (Martin et al. 2017), scaling it up for removing carbon from the atmosphere is considered to be highly speculative by a National Academy of Sciences study released in 2019 (NASEM 2019). The energy requirement for capturing carbon from the atmosphere (at concentrations of 419 ppm) is two to 10 times higher than capturing it from point sources, such as power plants, and will require more land area and higher capital costs, suggesting that it is unlikely to become attractive until carbon capture and sequestration (CCS) from these more concentrated sources is implemented (NASEM 2019). Estimated costs range between $30 to $1,000 per Mt of carbon captured with a median estimated cost of $345 (Martin et al. 2017; NASEM 2019). These costs do not include sequestration costs.

The world's largest direct air capture facility recently came online, increasing the world's direct air capture capacity by 40% (Hiar 2021). Built by the Swiss company Climateworks, the plant is expected to remove 4,000 metric tons of carbon from the atmosphere annually at a current cost estimated to be $600–800 per Mt. The Canadian firm Carbon Engineering Ltd. is planning direct air capture plants capable of capturing 250 times more carbon.

Figure 6.8. Conceptualization of direct air capture system (Carbon Engineering, Ltd.).

6.4.2 Methane Capture and Removal

Methane concentrations have increased significantly since the preindustrial period and trap 84 times more heat than carbon dioxide. Methane emissions are primarily from fossil fuel consumption and the agricultural sector. Recently, scientists have started to advocate for the removal of methane to slow down or reverse global warming. A recent study reports that reducing methane concentrations in the atmosphere by 40% can potentially reduce global temperatures by 0.4°C by 2050 (Abernethy et al. 2021). Generally, methane has a short atmospheric lifetime and by taking advantage of the natural oxidation of methane, researchers have proposed several methane removal methods. One option is to deploy photocatalysts across a large surface area to maximize and enhance methane oxidation under ultraviolet radiation. Like the flaring of natural gas at oil wells, the products of photocatalytic oxidation are water and carbon dioxide. Although carbon dioxide is an important GHG, its warming potential is much lower than the methane it replaces. A common photocatalyst is titanium dioxide which is used in paints. Applying thin layers of this photocatalyst paint on buildings or wind turbines could enable this reaction (Jackson et al. 2021; O'Grady 2021).

Alternatively, researchers have proposed ejection of iron-salt aerosols into the atmosphere to accelerate oxidation of methane. This reaction mimics the natural reaction of mineral dust particles in the atmosphere. One suggestion is to include iron additives in marine fuel oil to produce fine iron particles in the flue gas of the ships, which would oxidize methane in the atmosphere (Jackson et al. 2021; O'Grady 2021). Methane removal has only recently been proposed and is poorly developed compared to carbon dioxide removal technologies.

6.4.3 Bioenergy with Carbon Capture and Storage

Bioenergy with carbon capture and storage (BECCS) refers to a system that converts biomass into energy products and a concentrated stream of carbon dioxide suitable for carbon capture and sequestration

(CCS), as illustrated in Figure 6.9. Unlike CCS of carbon dioxide produced in fossil power plants, BECCS represents net carbon removal from the atmosphere because biomass contains carbon photosynthetically fixed from the atmosphere (IEA 2011). BECCS offers three pathways to remove carbon: gasification, post-combustion, and oxy-fuel combustion (Martin et al. 2017). Gasification converts biomass into a gaseous stream of carbon dioxide and hydrogen. The hydrogen is either used to produce fuels or to generate electric power while the carbon is sequestered in geological formations. Post-combustion BECCS burns biomass in air, while oxy-fuel combustion BECCS burns biomass in a pure oxygen stream. In both cases, energy is recovered from the hot flue gas, followed by removal and sequestration of carbon. Oxy-fuel combustion BECCS has the advantage of much higher concentrations of carbon dioxide in the flue gas, simplifying its capture, but comes with the added expense of an air separation plant to produce the pure oxygen.

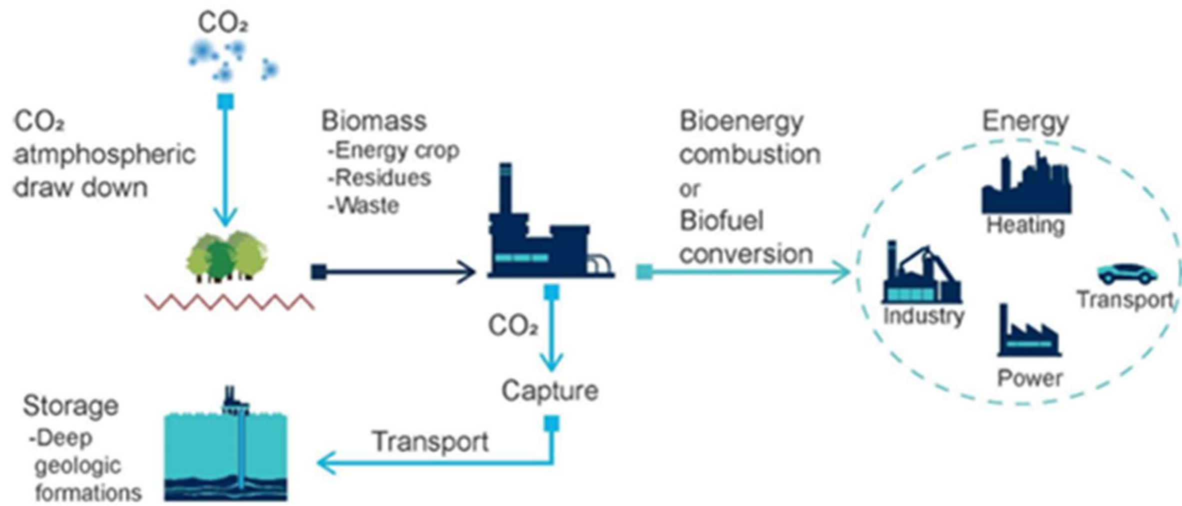

Figure 6.9. Schematic of bioenergy with carbon capture and storage (BECCS) process (Consoli 2019).

BECCS is currently in the development phase as biomass demonstration plants have yet to be constructed; however, the technological readiness is high because of the advancements in biomass gasification, combustion, and CCS strategies from integrated coal power generation (NRC 2015; Bui et al. 2021). Because some environmental groups raise land-use concerns about wide-scale deployment of BECCS, finding sites with both biomass resources and suitable geological formations for CO_2 sequestration can be challenging (Fridahl and Lehtveer 2018). However, BECCS remains a promising carbon removal pathway in agricultural areas where smaller BECCS plants can feed into hub locations for carbon dioxide transport (Donnison et al. 2020). The potential for carbon dioxide removal through BECCS in the United States is estimated between 0.5–1.5 gigaton of CO_2e per year, while estimated costs range between $20–$440 per Mt of carbon (median cost of $61; Martin et al. 2017; NASEM 2019). The type of BECCS plant, location, and biomass feedstock will impact the removal potential and process cost.

6.4.4 Capture and Sequestration of Biogenic Carbon Dioxide

Biogenic carbon dioxide that is a by-product streams from agriculture processing facilities can be captured and geologically sequestered for carbon removal (Fry et al. 2017). Corn ethanol plants are a prominent example of such facilities, with ethanol and carbon dioxide each representing one third of the mass yield. Iowa currently produces 30% of all ethanol in the United States, using more than 1.3 billion bushels of corn (ICGA 2021). It is estimated that up to 60% of the 45 million Mt of carbon dioxide produced each year by fermentation in biorefineries in the United States could be captured at a cost below $25 per Mt of carbon dioxide (Sanchez et al. 2018).

Sequestration of waste biogenic carbon dioxide has already been demonstrated in the Midwest by several projects that captured between 0.1 and 1.0 million Mt of carbon dioxide per year over the last decade (Kemper 2015). The Illinois Basin Decatur Project and subsequent Illinois Industrial CCS project captured and compressed carbon dioxide from an Archer Daniels Midland (ADM) ethanol plant and subsequently injected it into a nearby saline formation (Kemper 2015; NASEM 2019). The greatest challenge to adopting this technology is the need to build high-pressure pipelines to transport the carbon dioxide from ethanol facilities to underground sequestration sites. Fry et al. (2017) suggest that 9.85 million Mt of carbon dioxide per year could be captured across 34 ethanol plants and transported in a pipeline network to the Permian Basin. More recently, Navigator CO_2 Ventures LLC (2021) has proposed a 1,200 mi (1,931-km) pipeline connecting five Midwestern states toward injection points in Illinois (Fig. 6.10A). Summit Carbon Solutions (2021) similarly proposed a 710-mi (1,143-km) underground pipeline to transport carbon dioxide emissions from ethanol and other industrial agricultural plants to North Dakota for injection and permanent storage (Fig. 6.10B) (Eller 2021; Hytrek 2021). Both projects suggest carbon dioxide sequestration potential of up to 12 million M t per year.

A

B

Figure 6.10. Proposed (A) Navigator CO_2 Ventures LLC (Navigator CO_2 Ventures LLC 2021) and (B) Summit Carbon Solutions carbon dioxide pipelines from existing ethanol plants (Hytrek 2021 via Summit Carbon Solutions).

6.4.5 Geological Sequestration of Carbon Dioxide

Carbon dioxide capture from fossil fuel power plants or refineries for the purpose of reducing GHG emissions or from biopower plants or biorefineries for the purpose of net carbon dioxide removal from the atmosphere requires long-term storage of the gas for 100 years or longer (Fig. 6.11). This carbon capture and sequestration (CCS) requires capture of carbon dioxide emissions released from a process, compression or liquefaction of the carbon dioxide, transport to a storage location via pipeline, truck or ship, and injection into the subsurface (DOE/NETL 2010). Carbon dioxide can be stored underground in saline formations, depleted oil and gas fields, and unmineable coal seams (Table 6.4). Saline formations, composed of rocks containing salty water in their pore space, have by far the greatest potential for carbon dioxide storage by virtue of their vast extent. An estimated 2,379–21,633 billion metric tons of carbon dioxide could be stored in North America alone (DOE/NETL 2015). In general, CCS is considered safe and effective based on a foundation of extensive subsurface knowledge and experience from hydrocarbon development and exploration, validation of monitoring tools, and best practices developed through the Regional Carbon Sequestration Partnerships (RCSPs), and the results of numerous worldwide pilot and commercial projects (Damiani 2020).

In selecting a site for long-term sequestration in saline formations, the storage complex must meet several requirements. The formation must be at sufficient depths to avoid stratigraphic intervals containing drinking water. It must also be at a critical depth, approximately 800 m or 2,600 ft, for carbon dioxide to remain as a supercritical fluid (a thermodynamic state in which the carbon dioxide is neither distinctly gas or liquid). The storage site must furthermore have sufficient capacity to store large volumes. The capacity is determined by the amount of pore space in the formation and the formation permeability, which allows for the injection of carbon dioxide into the formation (DOE/NETL 2017). Containment of carbon dioxide in the formation is accomplished by the presence of a seal, overlying low-permeability layers that prevent the upward migration of carbon dioxide, and a trapping mechanism to prevent the lateral migration of carbon dioxide. Trapping mechanisms include stratigraphic trapping through lateral low-permeability within the rock formation and structural trapping where structural folding of the rock formation blocks the migration of carbon dioxide. The mechanisms that act to trap carbon dioxide in long-term sequestration are the same mechanisms that have trapped carbon dioxide in naturally occurring underground reservoirs as well as underground natural gas storage projects, acid gas injection projects, and naturally-occurring oil and gas reservoirs.

Over time, the carbon dioxide will migrate as a plume by dissolving in the saline water present in the pore space (Benson et al. 2005). Some volume will remain "behind" in the pore space via residual trapping. The carbon dioxide can also react with the rock formation to create solid carbonate minerals in a process called mineral trapping (Ajayi et al. 2019). The duration of containment of the carbon dioxide is estimated to be on the scale of at least 10,000 years (Alcalde et al. 2018). Leakage out of the storage site can occur through faults, injection points, or through abandoned wells, but is considered unlikely at well-regulated sites (Alcalde et al. 2018). The EPA has developed specific criteria for Class VI wells that permit sequestration of carbon dioxide, which involves testing and monitoring of the migration of the carbon dioxide plume over time (USEPA 2021d).

Existing commercial carbon dioxide storage projects are predominantly associated with gas production and located in an offshore setting (Benson et al. 2005). A total of 22 million Mt of carbon dioxide have been stored at the Sleipner and Snøhvit gas fields in the North Sea since 1996 (Damiani 2020). As of 2020, more than 11 million Mt of carbon dioxide have been stored in North America through projects associated with the Regional Carbon Sequestration Partnership Initiative Project, a nationwide network formed by the United States Department of Energy. The first project in the United States to drill an EPA Class VI well was at the Illinois Industrial CCS Project in 2017 where a total of 1.71 million Mt of carbon dioxide was successfully stored (DOE/NETL 2020).

Proposed projects to sequester carbon dioxide originating in Iowa involve transporting carbon dioxide via pipeline to North Dakota and Illinois. Summit Carbon Solutions' proposed 710-mile underground pipeline could store carbon in North Dakota at a rate of 10 million Mt per year. It is estimated that North Dakota has the potential to store between 76,000 and 252,000 million Mt of carbon dioxide (Western Dakota Energy Association 2021). A 1,200-mi pipeline project by Navigator CO_2 Ventures would transport carbon dioxide to south central Illinois.

The potential for carbon storage in Iowa was investigated by the Iowa Geological Survey as part of the Regional Carbon Sequestration Partnership Initiative Project. Their evaluation identified saline aquifers in southwestern Iowa at sufficient depth to render it potentially viable for storage (Witzke et al. 2018). Evaluating storage potential in Iowa is hindered by the lack of subsurface data that is otherwise common in areas of oil and gas development. During the exploration and development of oil and gas, subsurface data provide information on subsurface rock formations, structure, and geologic hazards. To understand the full potential of storage in Iowa, data acquisition in the form of subsurface seismic surveys, borehole geophysical logs, and core samples are needed.

Injection of carbon dioxide into geological deposits was originally developed for enhanced oil recovery, which involves injecting carbon dioxide into a depleted oil field to increase oil production. The injected carbon dioxide decreases the viscosity of the fluids in the formation and can increase oil recovery by as much as 18%. West Texas has utilized around 600 million Mt of carbon dioxide over several decades for enhanced oil recovery (Hill et al. 2013). Although use of captured carbon dioxide for enhanced oil recovery might seem contrary to the goal of removal of carbon dioxide from the atmosphere, some studies suggest that such projects can achieve net carbon negative status (Núñez-López and Moskal 2019). However, potential storage of carbon dioxide in oil and gas fields in North America is estimated to be only 186,000–232,000 million Mt tons, far less than estimates for saline formations (DOE/NETL 2015).

Table 6.4. Estimates of prospective carbon dioxide storage for North America in million Mt (DOE/NETL 2015).

	Low	Medium	High
Oil and Natural Gas Reservoirs	186,000	205,000	232,000
Unmineable Coal	54,000	80,000	113,000
Saline Formations	2,379,000	8,328,000	21,633,000
Total	2,618,000	8,613,000	21,978,000

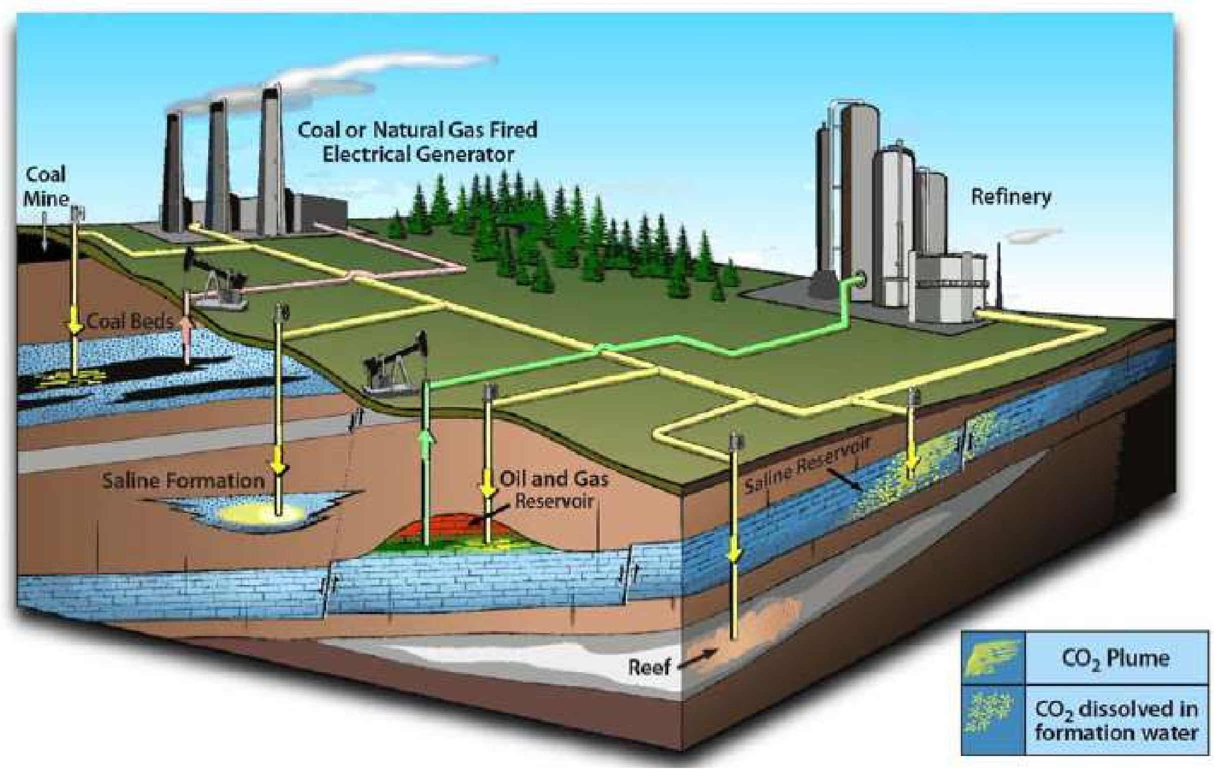

Figure 6.11. Schematic of carbon dioxide sources from industrial processes being stored in various geologic formations (DOE-NETL 2020). Pipeline color coding: yellow = carbon dioxide injection wells; green is oil and gas producing well; pink = gas producing well.

6.4.6 Pyrolysis of Biomass to Biochar

Pyrolysis is a moderately high temperature process (around 600°C) performed in the absence of oxygen that decomposes biomass into gaseous, liquid, and solid products. The gaseous and liquid products can be upgraded into energy products. The solid product, a form of charcoal commonly known as biochar, has received considerable attention for its ability to improve fertility of weathered and degraded soils (Laird 2008; Laird et al. 2009). Its recalcitrance to biological decomposition also makes it an ideal carbon sequestration agent. Like BECCS, pyrolysis of biomass and the sequestration of the resulting biochar achieve net carbon removal from the atmosphere, yet with the advantage of simpler sequestration compared to gaseous carbon dioxide and simultaneous ecosystem services (Fig. 6.12).

When biochar is applied to soil, about 10–40% of carbon is lost to mineralization over a few months. However, the remaining 60–90% of carbon is stable and has the ability to store carbon in the soil for thousands of years. As reported by many studies, the varying lifespan of biochar in the soil is primarily due to the quality of biochar, which is based on the configurations of the pyrolysis process (Laird et al. 2009). Biochar also has better recalcitrance characteristics when compared to plant or animal residues, suggesting that the application of biological residues only provides temporary carbon storage as opposed to the long-term carbon storage ability observed in biochar. Another source reports that biochar can capture and store carbon equivalent to 1.84 gigatons of carbon globally at a median cost of \$25 per Mt CO_2e (Martin et al. 2017). Additionally, biochar application to soil has the potential to increase crop yields by 38–45% and to decrease the utilization of fertilizers by 20% (Kung et al. 2013). This would decrease the cost and pollutant emissions associated with fertilizers.

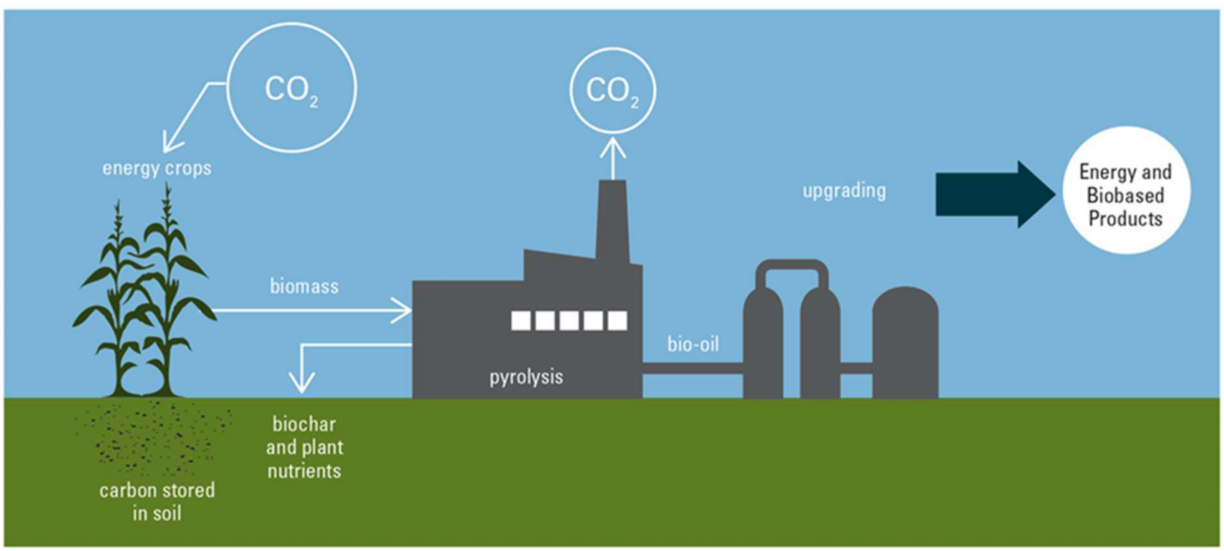

Figure 6.12. Pyrolysis of biomass to produce biochar for storage in soils.

6.4.7 Microbial Carbonate Precipitation

Carbon dioxide reacts with silicate in the environment, resulting in carbonate minerals (Mun and Cho 2013; Salek et al. 2013). Different approaches have been proposed to enhance carbon dioxide mineralization (CDM) as a method to sequester carbon from the atmosphere. They include **abiotic processes** (such as aqueous carbon mineralization in chemical reactors and injection of carbon dioxide into basalt rock formations) and **biotic processes** that can be described as microbial carbonate precipitation, which is particularly attractive for integration into Iowa agriculture. Two implementations of microbial carbonate precipitation are envisioned: in soils and in biotechnological processes.

For microbial precipitation in soils, fine powders of iron, calcium, and/or magnesium silicate rocks are incorporated into soils in a manner similar to liming of soils (Salek et al. 2013). Respiration in plant roots releases biogenic carbon dioxide to the soil in relatively high concentrations. Instead of migrating to the soil surface and being released to the atmosphere, some of the carbon dioxide reacts with water and the silicates to form bicarbonate ions (HCO_3^{-2}) and silicon dioxide (sand). Microorganisms in the soil precipitate the water-soluble bicarbonate ions as water-insoluble iron carbonate ($FeCO_3$), calcium carbonate ($CaCO_3$), or magnesium carbonate ($MgCO_3$) (Meysman and Montserrat 2017). Because the ultimate source of the carbon was the atmosphere, this process achieves net carbon removal (Fig. 6.13). The process is attractive because of its relatively low cost, letting natural processes in the soil achieve mineralization. The Intergovernmental Panel on Climate Change reports that these finely powdered silicates can reduce global agricultural emissions by 34–68% annually.

Researchers at Cornell University describe these finely powdered silicate minerals as “rock dust” and have applied them to agricultural land. Early stages of experiments report that the application of basalt rock dust (a by-product of mining and manufacturing operations) is more effective at capturing carbon than biochar and manure, as it was able to sequester two times the amount of carbon even under the driest soil conditions (Cosier 2021).

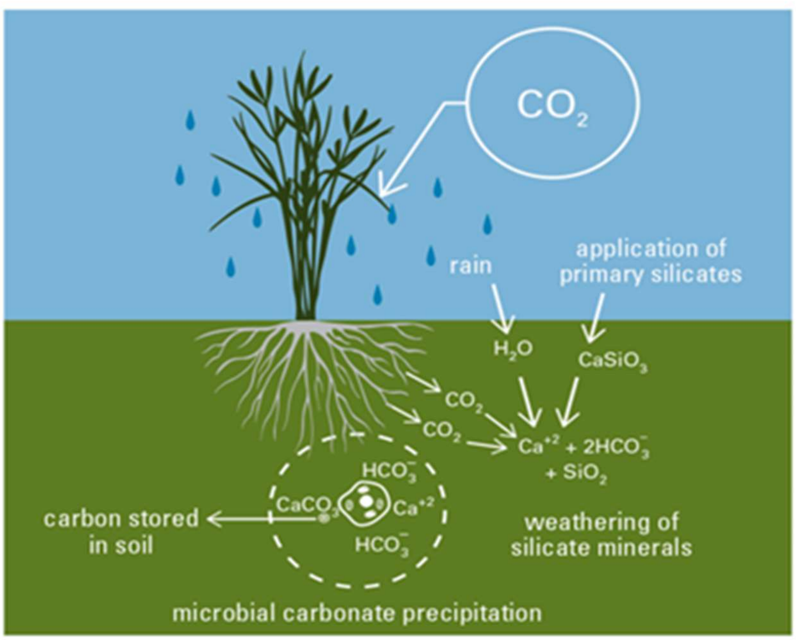

Figure 6.13. Microbial carbonate precipitation reaction in soils.

Rock dust is also known for improving soil quality and boosting crop yields. Researchers at Cornell also saw an increase of 30% in crop yield (corn and alfalfa) after the application of rock dust as it provides essential nutrients such as potassium and phosphorus. Additionally, rock dust could affect the nitrogen cycle which may result in reduced fertilizer usage (Cosier 2021). This has subsequent environmental and economic benefits. Researchers at Cornell are also looking into the availability and application of basalt on crops for the Midwest. Other studies have demonstrated the use of olivine, a type of mineral often used for this purpose because of its abundance and high dissolution rate. A carbon removal potential of 1.25 Mt carbon per ton of olivine has been demonstrated (Köhler et al. 2010). A report by the University of Michigan summarized the cost of this application as ranging between $29 and $157 per Mt CO_2e (Martin et al. 2017).

Microbial carbonate precipitation in biotechnological processes can also be envisioned, including biological nitrogen removal, bioelectrochemical systems, and anaerobic digestion (Zhu and Dittrich 2016). Anaerobic digestion is attractive because it generates large amounts of carbon dioxide in an environment conducive to microbial processes. The key requirement for mineralization is a sequence of acidity- and alkalinity-producing steps. In the case of anaerobic digestion, acetogenesis serves as the acidity-producing step while methanogenesis provides the alkalinity-producing step. In the presence of silicate minerals, bicarbonate ions arising from carbon dioxide generation can be precipitated as carbonates. The solid residues from the process can be recycled as fertilizers that provide nutrients to the soil. The removal of carbon as calcium carbonate also regulates the pH of the system and increases the methane content of the biogas (Salek et al. 2016).

Another favorable option is using biochar from pyrolysis in a system that combines microbial carbonate precipitate and anaerobic digestion (MCP-AD) of livestock manure with biomass (Fig. 6.14). Through microbial carbonate precipitate in anaerobic digestion, carbon can be sequestered economically, while solid residues from anaerobic digestion, which contain traces of biochar and other plant nutrients, can be applied to the soil to continue the sequestration of carbon, simultaneously improving soil quality. Combining these processes allows for both carbon removal and agricultural and livestock waste reduction, and it is also a more economic option as other energy products are being produced simultaneously. Similarly, this reaction can be integrated into existing plants, removing the need for an additional carbon removal unit (as required in chemical carbon mineralization), resulting in approximately 60% cost reduction per ton of carbon sequestered (Salek et al. 2013). On the other hand, biochar's high monovalent and divalent cation concentration can stimulate the carbonation reaction, while its porous structure and large surface area promote *in-situ* carbon removal (Shen et al. 2015). The addition of biochar in anaerobic digestion systems also enhances the conversion processes, resulting in higher biogas yield and increased process stability (Pan et al. 2019). A previous study showed that utilizing corn stover biochar in anaerobic digestion of sewage sludge resulted in an increase of 42% in biogas and of 54.9–86.3% in carbon removal (Shen et al. 2015).

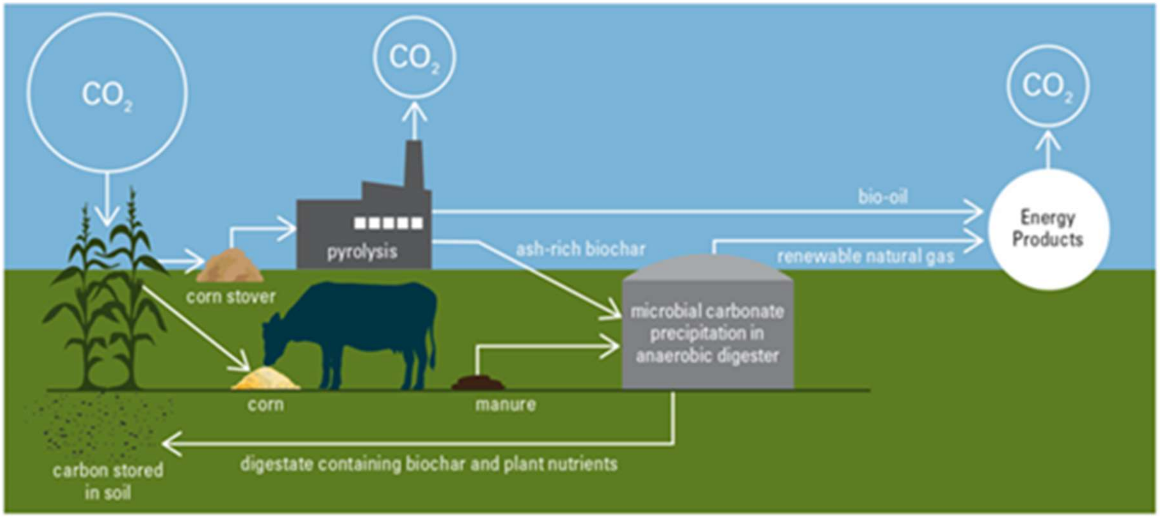

Figure 6.14. Microbial carbonate precipitation of biochar in anaerobic digestion.

6.4.8 Carbon Removal through Concrete Production

Concrete production is responsible for 8% of anthropogenic greenhouse emissions. The manufacturing process of concrete is very energy-intensive, particularly cement production, which emits about 600 kg carbon dioxide per ton of cement, representing 65% of the total GHG emissions from the process (Czigler et al. 2020). Cement is an essential material for concrete production and is produced via calcination. Through this process, high heat separates the carbon and part of the oxygen to form carbon dioxide, while the remainder forms calcium silicates that are cooled and milled into cement. Researchers have proposed that by taking advantage of this process, carbon dioxide can be captured and stored before it is released into the atmosphere. This captured carbon dioxide can also be used by other industries to make synthetic fuels. Alternatively, captured carbon dioxide can also be injected back into cement production, and by reacting with water, carbon dioxide is converted into calcium carbonate through mineral carbonation. This reaction not only improves the properties of concrete, making it stronger and able to withstand higher loads, but can also sequester 5–30% of carbon dioxide (Czigler et al. 2020; The Economist 2021). A Canadian company called CarbonCure has deployed this technology in over 400 plants globally. Furthermore, employing carbon capture and sequestration (CCS) at cement kilns is another option for GHG emissions reduction. Another approach is to replace cement with fly ash, a by-product from coal-fired power plants, or slag from steel and iron manufacture (Nature 2021).

An intriguing decarbonization option is to utilize low-cost biomass instead of coal to power the process and sequester all carbon dioxide emissions from the process. In this way, calcination of limestone becomes carbon neutral while combustion of biomass becomes carbon negative (The Economist 2021). Currently, the carbon dioxide captured at cement plants is at a capacity of 50,000–75,000 Mt carbon dioxide per year, while demonstration plants have the potential to capture 400.000–600,000 Mt carbon dioxide per year but are still under development (Tanzer et al. 2021).

6.5 Research Needs

This chapter identifies many engineering technologies for reducing GHG emissions and removing carbon dioxide from the atmosphere relevant to Iowa's participation in emerging carbon markets. It is not possible from the current state of knowledge to assess which of these engineering technologies has the most relevance to Iowa's present and future economy. Research could help improve technological readiness and lower financial risks associated with new technologies. Research that crosses agricultural, renewable fuels, and electrical sectors is especially needed, and would help Iowa take advantage of cross-sector synergies and position the state advantageously for a decarbonized future.

Research in the following areas could especially help close the substantial knowledge gaps that presently exist:

- Comprehensive techno-economic analysis, including financial benefit and risk assessment, of emerging low-carbon and carbon negative engineering technologies;
- Comprehensive life cycle assessment of emerging low-carbon and carbon negative engineering technologies based on data collected from field studies of Iowa agroecosystems and demonstration projects of relevant engineering technologies;
- Systems analysis to understand potential new synergies between agricultural and energy sectors enabled by carbon markets and renewable electric power. Prominent examples of emerging opportunities in advanced energy carriers include renewable diesel, sustainable aviation fuel, renewable natural gas, and green hydrogen and ammonia; and
- Demonstration projects to evaluate the potential benefit and technical and economic viability of engineering technologies in realistic environments and integrated with agricultural systems. Innovative partnerships among farmers, industries, communities, and Iowa's universities could move these efforts forward at a faster pace than if sectors were operating independently.

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Chapter 7. Agricultural Practice Adoption

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7.1 Highlights

- Iowa farmers are adopting agricultural practices that contribute to greenhouse gas (GHG) emissions reductions and soil carbon storage.
- Many farmers who are not currently using these agricultural practices either intend to try them in the future or are open to the idea of adopting them.
- Lack of knowledge, perceived inappropriateness, and cost relative to benefits are viewed as barriers to adoption.
- Further research is needed to examine potential relationships between practice adoption, carbon markets, and other incentives.

7.2 Background

Many common agricultural practices can contribute to reductions in GHG emissions and carbon removal, including tillage management (reduced till, no-till, strip-till), cropping practices (cover cropping, extended crop rotations, and diversification of cropping system including perennial crops), nitrogen management (nitrogen inhibitors, split applications, and in-season applications), and grazing management. Information on farmers' current use of such practices and potential willingness to adopt such practices can inform carbon program development and supporting policies.

Multiple social, economic, and ecological factors are associated with a farmers' decision to adopt these practices. These factors and their association with the agricultural practice adoption vary by the farmer (Upadhaya et al. 2020). Agricultural practice adoption is, thus, a complex concept and process, and understanding its complexity can help improve outreach and adoption. Practice adoption implies a commitment or expectation to continue using the practice in the long term. Farmers who have not used a practice typically start with a trial in a small part of the farm. Measured levels of adoption often include trialing, although it may not be possible to identify which farms are in trial mode. This creates a risk that long-term adoption may be overestimated. More broadly, adoption is often discussed as if it were a binary concept: adoption versus non-adoption. However, the farm-level adoption rate is a continuous variable in terms of the number of farmers who adopt a practice to any extent, from zero to 100%. For any farmer who is an adopter, a given practice may be adopted on any proportion of their land, from 0 to 100%. A practice may also be used in a field in only some years. For example, it may only be suitable for one of the crops within a rotation. Pannell and Claasen (2020) suggest a more nuanced and descriptive set of terms. “Full adoption” or “partial adoption” describe the extent of adoption within a farm. The “proportion of adopters” and the “extent of adoption” allow us to distinguish between the number of farmers who adopt and the aggregate area (or another measure of extent) over which they adopt. Adoption that varies over time within a given field can be referred to as “alternating adoption” (as opposed to "continuous adoption"). "Opportunistic adoption" relates to a practice that has been successfully trialed but is used only at times when suitable climatic or economic conditions exist. It is not permanently disadopted in the non-use years; it may just be set aside to wait for the right conditions.

We used multiple sources of data to estimate the current use and potential future adoption of agricultural management practices relevant to carbon management, including the (1) Census of Agriculture, (2) Iowa Nutrient Reduction Strategy tracking and reporting portal data (IDALS et al. 2021), (3) Iowa Farm and Rural Life Poll (IFRLP) surveys, (4) the Iowa Nutrient Reduction Strategy Farmer Survey (INRSS), and (5) the Iowa Department of Natural Resources Animal Feeding Operation database (Table 7.1).

Table 7.1. Data available on farmer adoption of or intention to adopt agricultural management practices.

Data Source (Citation)	Sample Size	Geographic Coverage	Temporal Coverage	Practices	Unit of Measurement	Remarks
Iowa Farm and Rural Life Poll (IFRLP 2021)	~1000	Statewide	2016, 2018, 2021	Multiple Practices	Use-Intention-Openness Scales	2021 Best Data for Estimating Intention, Openness
			2016, 2018	Multiple Practices	Acres	
			2020	Multiple Practices	5-point Increase-decrease scale	Adaptation Responses to Weather Variability
Iowa Nutrient Reduction Strategy Survey (INRSS; Nowatzke and Arbuckle 2018)	6006	Six HUC6 Watersheds	2015-2019	Multiple Practices	3-Point Use, Acres	
USDA Census of Agriculture (NASS 2014, 2019)		Statewide	2011, 2017	Cover Crops, Tillage Practices, Fertilizer Management, Land Use	Acres	In-Field Practices, Land Use
Survey of Agricultural Retailers (INREC 2020)		Statewide	2017-2019	Cover Crops, Tillage Practices, Fertilizer Management	Acres	In-Field Practices
Iowa Department of Agriculture and Land Stewardship Water Quality Wetlands Database (IDALS et al. 2021)		Statewide	2003-2019	Wetlands	Acres	Edge-of-Field Practices

Iowa Department of Agriculture and Land Stewardship Cost-Share Database (IDALS et al. 2021)		Statewide	2011-2019	Sediment Control Structures	Acres	Edge-of-Field Practices
USDA Natural Resource Conservation Service Cost-Share Database		Statewide	2011-2019	Sediment Control Structures	Feet, Number Installed	Edge-of-Field Practices
Iowa DNR Animal Feeding Operation Database (IDNR 2021a)		Statewide		Livestock and Poultry Facilities	Number, Types, Size, and Age	Animal Feeding Operations

Here, we first present estimates of the area, in acres, on which selected practices have been used in recent years. We secondly present survey data measuring the potential for Iowa farmers' future adoption of selected practices and potential barriers to adoption. It is important to note that these existing surveys and databases related to soil and water conservation and nutrient management, and that the practices were not directly presented as practices for carbon removal or carbon storage. While farmers' motivations to adopt practices may vary with their intended outcome, the barriers to adoption are likely to be similar. We thus expect these data can provide some understanding of farmer adoption and willingness to adopt carbon management practices.

7.3 Current Extent of Key Practices

7.3.1 Cover Crops

The United States Department of Agriculture (USDA) 2012 Census of Agriculture reported that approximately 370,000 ac of Iowa farmland (1.5 % of row crop acres) were planted with cover crops in fall 2011 (NASS 2014). By fall 2016, cover crops acres had increased to approximately 970,000 ac, or 4.1% of Iowa row crop acres, according to the 2017 Census of Agriculture (NASS 2019). The IDALS-led Survey of Agricultural Retailers estimated 1.6 million ac of cover crops the same year (Fig. 7.1). More recently, the Survey of Agricultural Retailers estimated that 2.2 million ac were planted in fall 2018. According to the county-level data from the 2017 USDA Census of Agriculture (NASS 2019), the eastern and southern regions of Iowa show the highest rates of cover crop use.

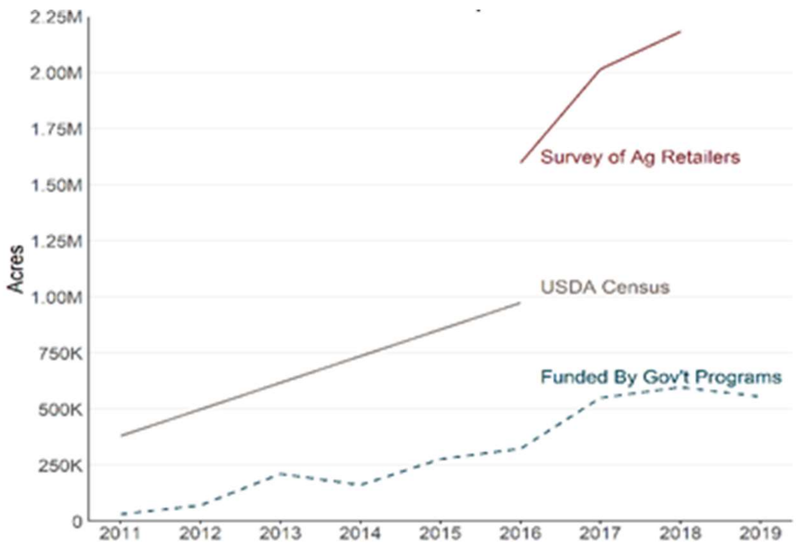

Figure 7.1. Annual cover crop acres in Iowa, according to the Survey of Agricultural Retailers and the Census of Agriculture. Acres funded by public government programs, including cost-share and Iowa's crop insurance program, represent a portion of the total acres planted annually in Iowa.

7.3 2 Residue and Tillage Management Practices

Residue and tillage management practices determine the amount, orientation, and distribution of crop residue on the soil surface year-round. No-till is a practice that minimizes soil disturbance and maintains most of the crop residue on the soil surface throughout the year. In 2012, the USDA Census of Agriculture estimated that no-till was used on about 30% of row crop acres planted (6.9 million ac) in the state (Fig. 7.2) (NASS 2014). According to both the Census and the Survey of Agricultural Retailers, since 2012, no-till acres have increased to approximately 8.2 million ac. No-till practices account for a higher portion of row crop acres in the rolling landscapes of western Iowa (i.e., the Loess Hills region) and some southern and northeastern watersheds (Fig. 7.3A).

Conservation tillage applies to any tillage system that covers 30% or more of the soil surface with crop residue after planting to reduce soil erosion. Conservation tillage was practiced on an estimated 8.8 million acres in 2012. Since then, conservation tillage has increased to approximately 10 million acres, according to both the Census of Agriculture and the Survey of Agricultural Retailers (IDALS et al. 2021). The use of conservation tillage is distributed across the state, with higher rates of use in the western, north central, and northeastern regions of Iowa (Fig. 7.3B).

The increased use of conservation tillage and no-till in row crop operations since the 1980s is paired with a decrease in the use of conventional tillage. According to the USDA Census of Agriculture, conventional tillage was used on an estimated 8 million ac during 2012 and has decreased to approximately 5 million ac in 2019 (IDALS et al. 2021).

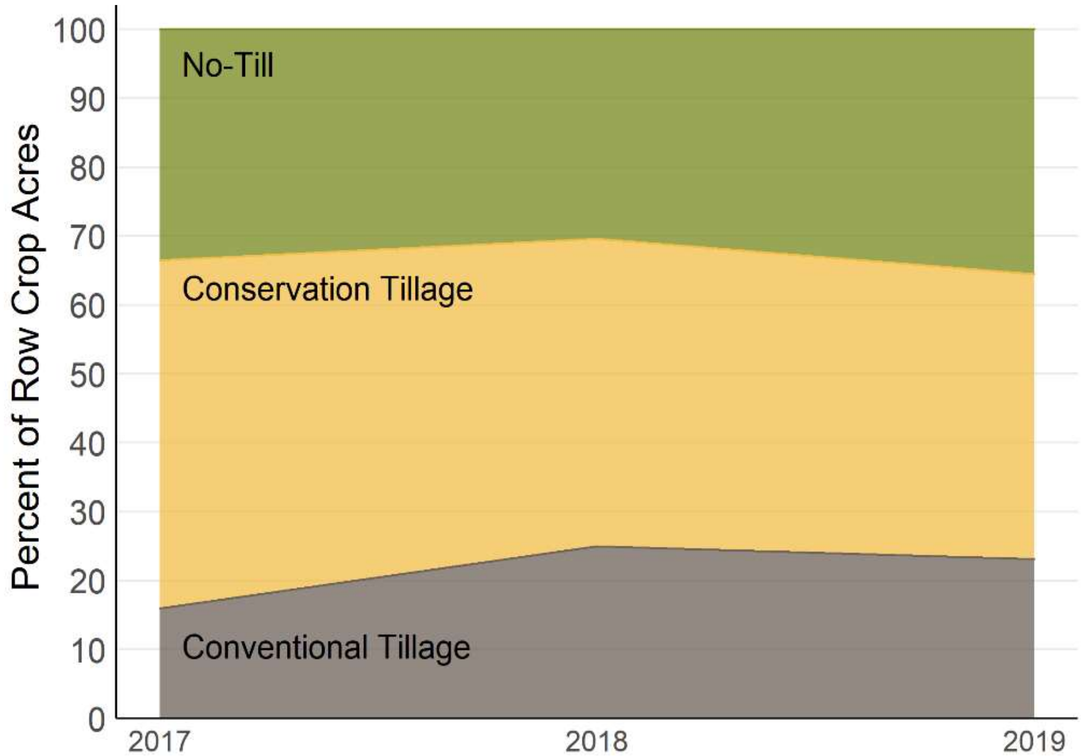

Figure 7.2. Acres of no-till, conservation tillage, and conventional tillage no-till over time, according to recent annual estimates derived from the Survey of Agricultural Retailers.

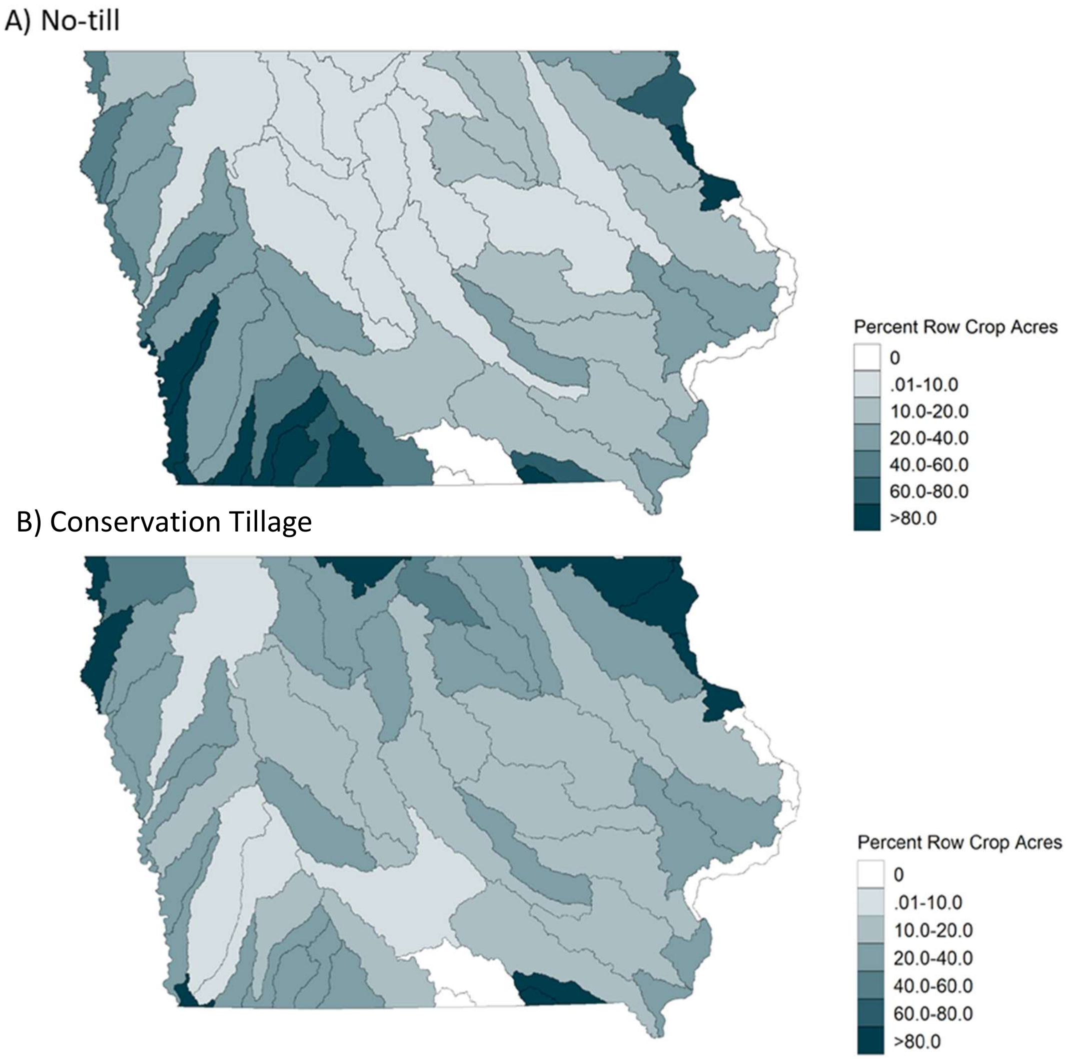

Figure 7.3. Spatial distribution of (A) no-till and (B) conservation tillage acres as percent of row crop acres, based on the 2017 Census of Agriculture.

7.3.3 Fertilizer Management

According to the 2012 Census of Agriculture, approximately 19.6 million ac of Iowa farmland were treated with commercial fertilizer, lime, and soil conditioners. Between 2012 and 2017, farm acres treated with commercial fertilizer decreased to approximately 19.2 million ac (NASS 2014, 2019).

The per acre application of commercial and manure nitrogen varies in corn-soybean rotations and continuous corn rotation. During the 2017–19 period, in corn-soybean rotations, corn acres received between 170–178 lb of commercial and manure nitrogen on average (IDALS et al. 2021). Continuous corn rotations, on average, received between 200–202 lb of commercial fertilizer per ac during that time. These annual nitrogen fertilizer rates represent statewide averages; however, nitrogen application rates to corn vary across agricultural fields and, in some cases, vary within a field. The percent of total acres that received different levels of commercial nitrogen rates varies by crop rotation. In 2019, for example, 40% of corn-soybean acres received 176-200 lb of commercial nitrogen in their most recent corn year, and 33% received 151–175 pounds. Some acres lay at the ends of this distribution, with 12%

of acres receiving 150 lb of nitrogen per ac or less and 15% receiving 201 lb per ac or more. There was a similar distribution for continuous corn rotations, with 51% of acres receiving 176–200 lb of commercial nitrogen fertilizer.

7.3.4 Manure Storage and Handling Facilities

According to the Iowa Department of Natural Resources (IDNR), there are 10,762 active animal feeding operations that contain 300 or more animal units.[15] IDNR maintains a database on facilities at or above this threshold (IDNR 2021a). About 9,796 of these are active confinement facilities, and 2,190 facilities are open lot. There are also 6,623 sites with a manure management plan that contain 500 or more animal units in confinement. There are 173 sites with a nutrient management plan that contain 1,000 or more animal units in open lots.

A majority of confinement facilities are swine production operations. There are approximately 8,769 swine confinement facilities, while 743 are beef and 206 are dairy facilities (Fig. 7.4). Manure facilities are of six major types based on storage structure (Table 7.2). According to IDNR, more than 7,669 facilities with 300 or more animal units have concrete storage structures, while 611 have earthen structures (Table 7.2). Concrete storages are generally deep pits under swine facilities, although some swine operations have separate concrete and slurry store storage.

There are only 12 anaerobic digesters that can be used to treat manure, crop residues, and multiple municipal and industrial organic wastes. According to IDNR, two of these 12 digesters in Iowa are permitted but have not yet been constructed.

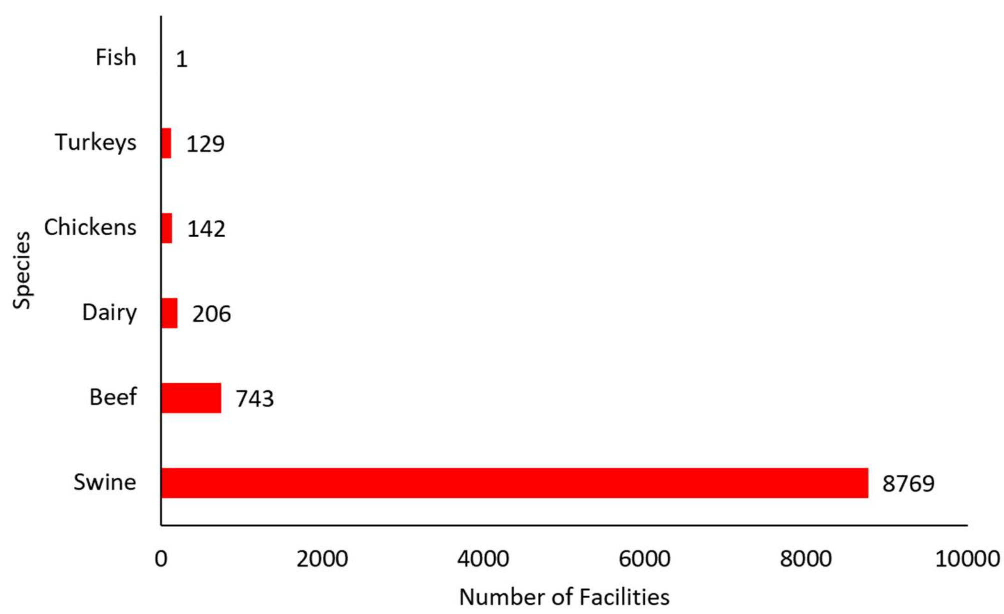

Figure 7.4. Active confinement facilities with more than 300 animal units by species.

[15] The IDNR calculates animal units "using a factor which converts animals of different species or sizes into equivalent units. Animal units for each species can then be added together to determine the capacity of a facility. The animal unit capacity and type of operation determine which state or federal regulations apply to that operation" (IDNR 2021b).

Table 7.2. Number of active facilities with 300 animal units or more with storage structure listed.

Facility Type	Number of Facilities
Earthen Storage Basin (ESB), Lagoons (Aerobic/Anaerobic)	611
Concrete Storage	7669
Slurry Store	235
Sand Settling	47
Settled Open Feedlot Effluent Basin	161
Alternative Treatment (AT) Systems (VTA/VIB)	28

7.3.5 Land Use

Iowa's total land area is 36 million ac, and according to the 2017 Census of Agriculture, nearly 90% of Iowa's total area is dedicated for agricultural purposes, with total agricultural land averaging 31 million acres since 1982 (NASS 2019). The land area dedicated to field crops—corn, soybeans, and other annual and perennial crops—has remained relatively steady since the 1980s, averaging 27 million ac. Since that time, there has been a steady and significant decline in pasture and hay acres. Acres enrolled in the United States Department of Agriculture Conservation Reserve Program, which aims primarily to convert environmentally sensitive land from crops to perennial cover, has fluctuated between approximately 1.5 and 2 million acres in Iowa since the start of the program in 1986 (IDALS et al. 2021).

7.3.6 Wetlands

Wetlands that are designed to improve water quality can achieve a reduction in nitrogen load of 52% (IDALS et al. 2021). As of 2019, Iowa had 95 wetlands designed specifically for water quality improvement (IDALS et al. 2021). Of these 95 wetlands, nine were installed in 2019 alone. Most of these wetlands have been constructed under the Conservation Reserve Enhancement Program (CREP). These wetlands have a cumulative drainage area of 124,000 ac. Additional wetlands are designed for other purposes, including flood mitigation, wildlife habitat, and cropland retirement. A project conducted by the Iowa DNR is currently accounting for and reporting the extent of these other types of wetlands.

7.3.7 Sediment Control Structures

There is extensive use of sediment control practices in Iowa's agricultural land, including terraces, farm ponds, water and sediment control structures, and grade stabilization. Currently, a significant proportion of sediment control practices are constructed with financial assistance from state and federal government cost-share programs. An estimated 217,000 ac are treated by terraces, water and sediment control basins (WASCOBs), ponds, and grade stabilization practices that have been installed under government cost-share programs since 2011 (IDALS et al. 2021). Owing to the topography and soils of the southern and northeastern regions of Iowa, erosion control practices are concentrated primarily in those geographic areas.

7.4 Potential Adoption of Key Practices

The degree to which Iowa's farms can reduce GHG emissions or remove carbon depends in large part on the agricultural practices and technologies that Iowa's farmers employ. This section examines current and, perhaps more importantly, potential future use of key carbon removal and storage practices. The literature on soil and water conservation practice adoption and related theory shows that openness to or intention to adopt are critical antecedents to adoption decisions and behaviors (Prokopy et al. 2019; Rogers 2003). Informed by this literature, the 2021 Iowa Farm and Rural Life Poll (IFRLP) survey, an annual survey of Iowa farmers, included a set of questions that measured both farmers' current use of key practices and their intentions and potential openness to practice adoption.

The 2021 IFRLP survey provided respondents with a list of 21 management practices that have the potential to reduce GHG emissions and/or remove carbon. The question regarding practice use was designed to measure current use among adopters, but also intentions of near-term adoption and openness to future use among non-adopters. The question set asked farmers to select one of four responses: "I used the practice in 2020"; "Not used in 2020 but intend to use within 3 years"; "Not planning to use within three years, but open to idea of future use"; and "Not used in 2020, no plans to use it." (Table 7.3). Because the practices listed are most applicable to row crop farmers (e.g., corn, soybean), the data in Table 7.3 are from the 86% of survey respondents who produced at least some row crops in 2020 (942 out of 1,095).

Table 7.3. Use, intention to use, and openness to use of selected carbon removal and reduction practices based on data from the 2021 Iowa Farm and Rural Life Poll (IFRLP); row crop farmer respondents only.

	I used the practice in 2020.	Not used in 2020, but intend to use within 3 years	Not planning to use within 3 years, but open to idea of future use	Not used in 2020; no plans to use it
In-field nutrient management practices	Percent			
Accounting for nutrient credits from different sources (e.g., manure, cover crops, soybeans)	69	10	12	10
Nitrogen stabilizers	59	10	14	17
Spring (starter) N application	46	7	19	28
Grid soil sampling for variable rate N application	43	15	21	22
Growing season N application (e.g., side-dress)	41	11	21	27
Determine N rate based on Corn N rate calculator (MRTN)	37	16	25	23
Soil health testing	34	22	28	17
Inject liquid N, P and K 6-8 inches underground	22	8	29	41

Use plant tissue testing to evaluate effectiveness of fertilizer program	16	20	34	31
Follow a formal, written nutrient management plan (e.g., NRCS 590 plan)	15	10	29	46
Edge-of-field practices				
Grassed waterways	83	4	4	9
Buffer/filter strips along streams or field edges to filter nutrients and sediment	61	5	14	20
Terraces	49	3	9	39
In-field buffer/filter strips (e.g., contour buffer strips) to filter nutrients and sediment	42	5	22	30
Water and sediment control basin (WASCOB) with stable outlet	20	6	25	49
Created or restored wetlands for nutrient removal	10	5	22	64
Prairie strips conservation practice	5	6	25	64
Soil management practices				
No till (all years of rotation)	48	9	17	26
Cover crops	31	21	25	23
Strip till or similar minimum disturbance tillage	25	9	23	43
Adding another crop to your crop rotation in addition to corn and soybean (e.g., alfalfa, small grain, other)	24	8	24	44

Among in-field nutrient management practices, the most commonly used were accounting for nutrient credits from different sources (69%), nitrogen stabilizers (59%), and spring nitrogen application (46%) (Table 7.3). Use of other practices ranged from 43% for grid soil sampling to a low of 15% for use of a formal nutrient management plan. Among non-users, large proportions of farmers either intended to use practices in the next three years or were at least open to the idea of using practices in the future. The proportion of farmers who had no plans to use these practices ranged from 10% for nutrient credits to 46% for formal nutrient management plans.

The most commonly reported edge-of-field practices were grassed waterways (83%), followed by buffer strips along streams or field edges (61%), and terraces (49%) (Table 7.3). Use of other practices ranged from 42% for in-field buffer or filter strips to a low of 5% for use of the prairie strips conservation practice. Among the non-users of the lesser-used practices, 31% of farmers were open to the use of WASCOBs, followed by 28% for in-field buffer strips, and 27% for nutrient removal wetlands. The proportion of farmers who had no plans to use these practices ranged from 9% for grassed waterways (compared to 83% already using) to 64% for nutrient removal wetlands (compared to 10% already using).

The most commonly used soil management practice listed in the survey was no-till (48%), followed by cover crops (31%), strip-till or similar “minimum disturbance” tillage (25%), and extended rotations (24%) (Table 7.3). The proportion of farmers with near-term intentions to try practices or potential willingness to try them in the future ranged from 46% for cover crops to 31% for prairie strips, indicating substantial possibilities for future adoption. That said, significant numbers of respondents reported no plans for future use of any practice.

Considering the results for current and potential practice use, a finding that stands out is the fact that the proportion of farmers who either already intend to try practices or are open to the idea was relatively high for many practices. Further, the set of questions did not mention potential financial assistance or remuneration. It is likely that if a scenario in which some form of payment had been offered, such as from carbon markets, intention and openness would have been higher.

7.5 Barriers to Practice Use

A number of studies have examined barriers to adoption of conservation practices (e.g., Prokopy et al. 2019, 2008; Ranjan et al. 2019) and have shown that a diverse range of economic and agronomic factors, social norms, perceptions of government programs, farm characteristics, land tenure factors, and knowledge-related factors can be major barriers to conservation adoption (Nowatzke and Arbuckle 2018; Prokopy et al. 2008, 2019; Ranjan et al. 2019). A better understanding of the barriers that farmers face can help to advance programs and policies meant to surmount challenges. Both the INRS and the IFRLP farmer surveys have explored perspectives on barriers to practice use.

The INRS survey contained a set of questions that asked non-users of 12 key practices to indicate if any of five potential barriers applied to them with respect to each of the practices. The five types of barriers that are commonly cited were developed in consultation with major agricultural stakeholders: “risk to crop yield,” “cost too high compared to benefits,” “don’t know enough about it,” “not appropriate for the farm’s soil or terrain,” and, “rented land: landlord not supportive or investment not worth it” (Table 7.4).

Table 7.4. Barriers to use of selected practices, based on data from the 2015–2019 Iowa Nutrient Reduction Strategy Survey (INRS); row crop farmer respondents only.

	Risk to crop yield	Cost too high compared to benefits	Don’t know enough about it	Not appropriate for my farm’s soil or terrain	Rented land: landlord not supportive or investment not worth it
In-field nutrient management practices					
Spring nitrogen application	13	12	5	18	2
Growing season nitrogen application (i.e., side-dress)	8	17	11	21	2

Variable rate N application	4	22	28	10	1
Nitrogen rate based on Corn N Rate Calculator (MRTN)	4	7	52	6	1
Nitrogen stabilizer (e.g., N-SERVE)	2	37	18	8	1
Edge-of-field practices					
In-field buffer strips (e.g., contour) to filter nutrients and sediment	4	12	18	37	7
Buffers along streams or field edges to filter nutrients and sediment from runoff	4	15	17	33	6
Bioreactor(s)	2	16	51	15	4
Soil management practices					
Cover crops	12	38	32	12	9
No till (all years of rotation)	30	8	12	34	4
Strip tillage	6	17	24	31	3
Extended rotations (3 or more crops over a 3-5 year rotation)	7	19	20	19	6

Risk to crop yield was cited as a barrier by relatively few farmers across all practices. As would be anticipated, this barrier was selected more for in-field nutrient management and soil management practices than for edge-of-field practices. The practice most associated with the yield risk barrier was no-till, with 30% of respondents selecting it, followed by 13% selecting it for spring nitrogen application and 12% citing it as a barrier to cover crops use. For all other practices, fewer than 10% of farmers selected yield risk as a barrier to adoption (Table 7.4).

Compared to yield risk, “costs too high compared to benefits” was more commonly cited as a barrier to practice use. Thirty-eight percent of farmers indicated that it was a barrier to cover crop adoption, with a similar proportion (37%) citing it as a barrier to use of nitrogen stabilizers. Substantial percentages selected costs compared to benefits as a barrier to adoption of variable rate nitrogen (22%), extended rotations (19%), and growing season nitrogen (17%).

Lack of knowledge emerged as the most commonly cited barrier across practices. More than half of respondents indicated that lack of knowledge was a barrier to the use of a Corn Nitrogen Rate Calculator (52%), and bioreactors (51%) (Table 7.4). Substantial percentages cited knowledge barriers to adoption for cover crops (32%), variable rate nitrogen application (28%), and strip tillage (24%). Lesser, but still significant proportions of farmers selected lack of knowledge as a barrier to the remaining practices, with only one practice, spring nitrogen application, selected by fewer than 10% of respondents.

The barrier “not appropriate for my farm’s soil or terrain” was the second-most cited barrier to practice use. The highest proportions of farmers selected this as a barrier to in-field buffer strips (37%), no-till (34%), riparian or field-edge buffers (33%), and strip tillage (31%) (Table 7.4). It is important to note that this is a measure of “perceived” appropriateness. Although we do not know from the survey whether the practice would actually be appropriate, given that many of the practices listed would be appropriate for nearly any farm operation with row crops, it may be that some farmers have an inadequate understanding of how some practices might fit into their operations.

The final barrier listed, “rented land: landlord not supportive or investment not worth it,” was the least-cited barrier to the practices listed. Cover crops, at 9%, was the most selected practice under this barrier. Thus, fewer than 10% of farmers saw rented land as a barrier to use of any of the 12 practices.

In summary, the INRS survey data indicated that lack of knowledge, perceived inappropriateness, and cost relative to benefits were viewed as barriers to use of best management practice by a substantial number of farmers. It is important to note, however, that the survey provided only these five potential barriers. There are many more potential barriers (e.g., social norms) that were not measured. That said, these findings point to a need to target knowledge and economic barriers to facilitate future adoption of practices.

Whereas the INRS survey focused on barriers to specific practices (Table 7.4), the 2020 IFRLP survey examined more general perceived barriers to improvements in soil and water conservation (Table 7.5). The set of seven questions focused on varied dimensions of economic and agronomic barriers and was preceded by the text, “State and federal governments, land grant universities, and farmer groups have been promoting soil and water conservation practices for decades. Despite these long-term efforts, Iowa still has soil erosion and water quality impairment issues.” Respondents were asked to rate their agreement or disagreement on a five-point agreement scale.

Table 7.5. Farmer perspectives on reasons for continued soil erosion and water quality impairment issues, based on data from the 2020 Iowa Farm and Rural Life Poll (IFRLP); row crop farmers only.

	Strongly Disagree	Disagree	Uncertain	Agree	Strongly Agree
Economic barriers					
Pressure to make profit margins makes it difficult to invest in conservation practices.	2	14	16	56	12
Many farmers don’t have the economic resources to adopt sufficient conservation practices.	3	20	26	45	6
Concern that conservation practices (e.g., cover crops, no-till) might cause yield declines that would negatively impact Actual Production History (APH) and subsequent crop insurance and other USDA payments is a barrier.	5	23	32	35	6
There is not enough cost-share and other support available from government agencies.	3	18	41	31	7
Agronomic management barriers					
Soil erosion is difficult to avoid in corn-soybean production systems.	6	41	13	35	5
Nutrient loss is difficult to avoid in corn-soybean production systems.	5	38	21	33	4

Nutrient loss is difficult to avoid in tile-drained fields.	7	38	33	19	3

Among the economic barriers, the statement, “Pressure to make profit margins makes it difficult to invest in conservation practices,” received the most endorsement, with 68% percent of farmers in agreement (Table 7.5). More than half of respondents agreed that many farmers lack the economic resources to adopt sufficient practices (51%), and 41% agreed with a statement about concerns that practices could negatively impact yield history and subsequent crop insurance payments. The lowest-rated economic barrier item was, “There is not enough cost-share and other support available from government agencies,” with 38% in agreement but the plurality (41%) selecting the “uncertain” response category. The three agronomic management-related barriers garnered slightly greater disagreement than agreement.

7.6 Research Needs

There is an immense potential for developing the supply side of carbon credits in agriculture because there is an opportunity to improve the adoption of agricultural practices that have the potential to reduce GHG emissions and/or remove carbon. Being able to quickly establish additionality and monitor permanence is crucially important to developing credible carbon credits. However, for most agricultural practices that have the potential to reduce GHG emissions or remove carbon, there are insufficient data to accurately estimate and track adoption over time. Integrated measurement of the extent of adoption of different practices over time is needed to fill the critical data gaps that limit the ability to better inform the development of credible carbon credits in Iowa. Integrated measurement systems might make use of remote sensing, machine learning, and artificial intelligence. Further investment in these processes could automate monitoring processes and reduce the cost of administering carbon programs (see **Chapter 3** for more information).

This chapter explored the current extent of use, potential for future use, and barriers to the adoption of many agricultural practices. However, data specifically on motivations for and barriers to adopting practices that can reduce GHG emissions and remove carbon and participate in carbon markets are not available for Iowa. The state of Iowa could encourage universities to conduct comprehensive assessments of contextual factors that may influence farmers’ willingness to adopt practices and participate in carbon markets. Specifically, detailed understanding of farmers’ and livestock producers’:

- Past and current adoption of different practices that contribute to soil carbon and GHG management,
- Potential willingness and capacity to adopt different practices and participate in carbon markets,
- Structural or educational needs to adopt practices and participate in carbon markets, and,
- Willingness to accept payment for a suite of agricultural practices to reduce GHG emissions and remove atmospheric carbon dioxide.

The final item is needed to estimate the extent of the supply of agricultural credits in Iowa.

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Chapter 8. Agricultural Carbon Planning

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8.1 Highlights

- Currently available data and decision tools can be leveraged to assist farmers and advisors in carbon market planning.
- Landscape- and field-level carbon management data are needed to increase the credibility and utility of planning tools.
- Integration of data and tools used in planning and measurement, reporting, and verification (MRV) would enhance the efficiency of agricultural carbon management.

8.2 Background

Carbon management is a new opportunity for Iowa farmers and their agricultural advisors. The application of data and planning tools can provide farmers and their advisors with increased information and certainty in their decision-making processes and highlight potential opportunities for agronomic and environmental synergies—including for carbon management. Farmers and agricultural advisors have long harnessed data and planning tools to manage other on-farm and off-farm resources effectively and efficiently (e.g., nutrients, soil moisture and water, crop yields, etc.). Existing data and planning tools could be adapted and expanded to effectively support agricultural carbon planning and management in Iowa.

At landscape scales, carbon management via the application of conservation practices is more dependent on the overall scale of application, site conditions, and long-term collective management than upon explicit spatial considerations. Quantifying carbon management outcomes and other desired environmental benefits (e.g., water quality and landscape level habitat) that can be co-produced require varying degrees of spatially dependent hydrologic and landscape connectivity (Tomer and Locke 2011; Lovell et al. 2021; Rastandeh et al. 2021). To attain this connectivity requires planning that makes use of state-of-the-art geospatial planning data and tools to provide the necessary hydrologic and landscape analysis (Jager and Kreig 2018; Zimmerman et al. 2019). The geospatial findings of tools at basin scales can be combined with field-level conservation decision support tools that aid in identifying practices that would be most effective and suitable for a given site, at the lowest cost for landowners (e.g., Tomer et al. 2013; McLellan et al. 2018). Likewise, currently available landscape planning tools are fully compatible with field-level carbon assessment tools, such as COMET-farm (and other carbon calculators), for landowners and their advisors to estimate past, current, and future carbon and GHG dynamics.

Relative to carbon planning and management, the practices listed in Table 8.1 have varying degrees of spatial determination at field scales. Soil type, land use history, current management, moisture conditions, aspect, and other factors mediate carbon and GHG dynamics in and around any conservation practice. Perennial practices often can be applied on low-yielding or environmentally vulnerable farmland in ways that can lower whole-field production costs via increased input efficiency and increased average yields (Brandes et al. 2016). Some in-field conservation practices that farmers use to minimize erosion and nutrient loss can be broadly applied across highly variable site conditions (e.g., tillage, cover crops, land retirement via CRP). There are, however, many structural practices, including most perennial vegetation systems designed to mitigate field runoff related concerns (grass waterways, prairie strips, riparian buffers, water and sediment control basins), that involve more finely explicit spatial considerations associated with field-level topography, runoff patterns, and landscape positions (Liu et al. 2017). Structural practices that involve subsurface drainage, such as controlled drainage, wetlands, bioreactors, and saturated buffers, are explicitly a spatial function of hydrology, drainage, and land use.

There are also synergistic outcomes relative to carbon management and generating multiple benefits. For example, as discussed in **Chapter 4**, increased soil organic matter contributes to increased pore space and stable aggregate structure in soils, thus improving soil water and nutrient storage capacity, which at scale facilitate flood control and protect water quality. In-field practices, for example nutrient management, controlled drainage, conservation tillage, and cover crops, are opportunities for carbon management—and other important managed outcomes, such as soil moisture management, reduced erosion, and increased nutrient use efficiency (see **Chapter 4**). Edge-of-field practices, such as filter strips and riparian and saturated buffers, offer carbon management benefits associated with perennial vegetation—and are typically designed to intercept field runoff or subsurface drainage water to minimize nutrient and sediment loss from fields by slowing runoff, increasing infiltration and water storage, and by providing wildlife habitat.

When field-level practices are planned and located strategically at basin scales to connect to watershed hydrology and target specific pathways for nutrient and soil loss, the benefits can aggregate to scales beneficial to society (McLellan et al. 2018). For example, alternative cropping systems or land set-asides that involve perennials often have either commodity production objectives (e.g., feed, bedding, biomass) or broad local or regional conservation goals such as habitat. Furthermore, many practices directly contribute to terrestrial and aquatic habitat by creating habitat diversity, enhancing landscape connectivity, and protecting surface water (McConnell and Burger Jr. 2018). Table 8.1 summarizes various conservation practices that have the capacity to jointly contribute to multiple benefits, including carbon management. It is important to recognize that the ability of these practices to positively impact water quality, habitat, and carbon dynamics varies considerably from site to site and depends on location within a farm and watershed, scale of application, maturity of practice, site-level conditions (such as hydrology, soil, cropping history), weather, practice design, management characteristics, farmer experience, and availability of technical services.

Table 8.1. Agricultural and conservation practices and their multidimensional environmental effects. CO_2e = carbon dioxide equivalent.

Practice	Impacts of Practice on CO_2e	% Nitrogen Load Reduction[a] (Standard deviation)	% Phosphorus Load Reduction (Standard deviation)	Impacts of Practice on Habitat (Direct)[b]

Cover crops [c]	Current scientific knowledge:[d] • Likely to increase soil carbon, but dependent on other factors (e.g., tillage, soil texture and initial soil carbon, species, soil depth considered, and timeframe). • Effect on nitrous oxide emissions may be neutral.	31 (29)	29 (37)	Weak direct impacts
Drainage water management	Current scientific knowledge: • Drainage affects soil carbon stocks and emissions of nitrous oxide and methane. • As drainage systems are upgraded and expanded, they will affect both soil carbon stocks and nitrous oxide emissions. • Nitrous oxide emissions may decline.	33 (32)	--	No direct impacts
Reduced and no-tillage	Current scientific knowledge: • Reduced tillage slows the rate of soil organic matter decomposition, resulting in increased soil carbon storage.	--	33 (29) 90 (17)	No direct impacts
Bioreactor	Current scientific knowledge: • Bioreactors produce nitrous oxide and methane; nitrous oxide produced is less than that produced by crops • Optimizing hydraulic residence time and mass nitrate removal can be used to limit GHG production.	43 (21)	--	No direct impacts
Conservation cover[e]	Current science knowledge: • Living roots and above-ground biomass year-round prevent soil loss via erosion. • Deeper, more expansive root systems than annuals disproportionately increase soil organic carbon.	85 (9)	75	Overall moderate direct impacts with scientific consensus: • Successional habitat suitable for nesting by farmland birds • Corridors to facilitate animal movement between fragmented patches

	• After establishment, perennials require much less disturbance compared to cropland.			• Pollinator habitat and food resources
Contour buffer strips (prairie or vegetative)	See conservation cover	91 (20)	58 (32)	See conservation cover
Grass waterways	--	--	--	Weak direct impacts
Nutrient removal wetlands[f]	Current scientific knowledge: • Relative balance between net carbon removal and methane emission varies widely among wetlands and is sensitive to water regime. • Carbon removal approximately offsets methane in most temperate wetlands.	52	--	Overall strong direct impacts with scientific consensus: • Wintering and breeding habitat for several species of birds, including game birds • Fish, shellfish habitat and spawning areas
Riparian buffers (herbaceous,[g] forested[h])	Current scientific knowledge: • Likely to increase soil organic carbon over time • Lower nitrous oxide and methane emissions than adjacent crop fields	91 (20)	58 (32)	Overall moderate to strong direct impact with scientific consensus: • Successional habitat suitable for nesting by farmland birds • Corridors to facilitate animal movement between fragmented patches • Pollinator habitat and food resources
Saturated buffers	Current scientific knowledge: • Likely to increase soil organic carbon over time • Lower nitrous oxide emissions than adjacent crop fields	50 (13)	--	No direct impact unless a new riparian buffer is established
WASCOBs	--	--	85	No direct impacts

[a] A positive number reflects a load reduction, and a negative number is increased load. Data presented here comes from Thompson et al., (2019).

[b] Summary based on Iowa State University Conservation Learning Group (2020). Habitat impacts are highly variable and ultimately dependent upon species, maturity of practice, scale of application, landscape features and landscape positions, and practice management.

[c] Assumes cereal rye (*Secale cereale*).

[d] Based on information presented in Chapter 4.

[e] Conservation cover is any permanent vegetation cover. Here, the assumption is that the cover consists of native warm season grasses and forbs. Data presented is for CRP land retirement land uses.

[f] From Zhang et al. (2020).

[g] Assumes buffer consists of native warm season grasses and forbs.

[h] Assumes buffer consists of native of trees, shrubs, and warm season grasses and forbs.

Importantly, it has been noted that landowners and farmers who are inclined to integrate conservation practices into their farm systems have preferences for agricultural land use that feature economies of environmental scope (e.g., singular land use that jointly produce multiple environmental outcomes), particularly when there is deliberate policy supporting these outcomes (Wossink and Swinton 2007). The cost effectiveness of a singular conservation-oriented practice increases when there are multiple environmental benefits simultaneously produced (Wortmann et al. 2011; Pennington et al. 2017). Thus, there are economic metrics in favor of and social preferences for policy that facilitates multiple benefit, multi-scale land use (Gardali et al. 2020). Multiple outcome land use requires landscape or watershed scale planning and concomitant economic incentives that together 1) encourage adoption of practices where they are needed, 2) foster long-term landowner commitment to those practices, and 3) engender innovative market development (Reside et al. 2018; Nguyen et al. 2021).

The direct and opportunity costs of agricultural conservation practices that contribute to carbon management are an impediment to adoption and long-term maintenance of the practice (Rejesus et al. 2021). Across all practices, four primary categories of direct cost are relevant: (1) adoption lead-in costs (e.g., learning time); (2) costs associated with design, site preparation, and establishment; (3) costs of short- and long- term management, maintenance, and/or practice replacement; and (4) annual overhead costs. For relevant practices that remove land from production, opportunity cost of land is also relevant. There may also be indirect costs associated with certain conservation practices that are based on changes made to the cash crop system (such as needing to utilize integrated pest management or increased equipment use and/or modification). Finally, there are also transaction type costs associated with public or private conservation or carbon program participation. While there can be private benefits associated with various conservation practices and carbon management, they typically are not higher than implementation costs in the short run (Rejesus et al. 2021). For practices that remove land from production (e.g., riparian buffers, contour buffer strips, nutrient removal wetlands), private benefits may not compensate for long-term opportunity costs (Roley et al. 2016). All conservation benefits require a long-term commitment from landowners, and innovative and multiple (public and private) finance opportunities will be required to create temporally connected incentives and landowner buy-in.

Because of the public nature of conservation benefits at scale, USDA Conservation Reserve Program (CRP) and Environmental Quality Incentives Program (EQIP) programs are available to help offset private adoption costs of agricultural carbon management practices. Yet, the data are fairly clear that most acres enrolled in USDA-supported conservation programs return to agricultural production at the conclusion of contracted periods (Bigelow et al. 2020). A portfolio of land use incentives that promote long-term carbon maintenance and other environmental outcomes is often required to bridge short- and long-term interests. Public subsidies (e.g., via USDA conservation programs) may be available to offset private costs for establishing carbon relevant practices, as well as initial management and opportunity costs of land use (e.g., CRP land rent); yet, beyond contract periods, supplemental income may be needed to offset long-term opportunity costs associated with maintaining a practice in perpetuity (e.g., Clayton 2019; Gewin 2019).

As discussed in **Chapter 2**, carbon markets pay for both carbon storage in soils and reductions in GHG emissions. Involving carbon management with other planned environmental outcomes, such as water quality, that also require sustained investment and management may strengthen investment motivations and lead to multiple revenue streams for landowners, thus, providing a bridge between short- and long-term financial needs and hedge against potential volatility in any one revenue stream.

The remainder of this chapter covers calculators and planning tools to support resource decision-making (i.e., carbon management, other environmental benefits) and a case study highlighting potential opportunities for co-production of carbon and water quality benefits. The chapter concludes with a discussion of high-priority research needs to support and expand agricultural carbon planning in Iowa.

8.3 Calculators and Planning Tools to Support Resource Decision Making

There are a number of different farm-to-watershed scale calculators and planning tools and for-fee planning services available to landowners to support agricultural carbon planning (e.g., *Ag Leader Technologies, Continuum Ag—Topsoil, Indigo Carbon, Truterra, etc.)*. Here, we summarize the science of calculators and planning tools and provide examples of five calculators and two planning tools.

8.3.1 Calculators

Calculators are modeling tools created to educate, evaluate, and support decision-making. Calculators provide a generalized estimate of the carbon and GHG emissions related to the management scenario set by the user and may allow for rapid appraisals. They are more accessible to a wide range of audiences (e.g., environmentalists, farmers, policy makers, planners, students, the general public), and do not require substantial modeling expertise to run and interpret. While they are more straightforward to use than process-based models (covered in **Chapter 9**), they often still require hours to generate a reasonable prediction.

Calculators are frequently based, at least in part, on process-based models as well as estimates of GHG emissions or soil organic carbon (SOC) storage based on guidance provided by the Intergovernmental Panel on Climate Change (IPCC). Some tools concentrate only on a given land use type—for example, forestry, agriculture, pastoral systems—whereas other tools are very comprehensive and may integrate both. Although many calculators lack a spatial interface, they represent carbon and GHG dynamics for a particular geography, whether it be a single point (e.g., a farm field), across landscapes and regions, or for the entire globe. In addition to representing an agroecosystem, many GHG calculators include carbon emissions from energy use through transport and nitrogen fertilizer, which is not directly possible with process-based agroecosystem models.

At present, this is an active area of research in which existing models are being improved and new models are becoming available. Over the past 10 years, the number and scope of calculators in particular have increased, demonstrating the growing need by different entities to understand their carbon footprint. Studies by Colomb et al. (2013) and Peter et al. (2017) revealed more than 36 tools, although only 18 were covered in detail. While the number of tools available allows users to select the best calculator for their purposes, it creates a selection dilemma for those with limited expertise: there are currently no standards or conceptual frameworks to guide the selection of the most effective calculators. Some tools, but relatively few overall, have been published in peer-reviewed scientific journals.

While many calculators are available, some have received more substantial development than others. The five calculators detailed below and in Table 8.2 have received substantial development and are currently being used by carbon programs in Iowa. No comprehensive resource is available that covers and compares all calculators.

Table 8.2: Summary of carbon and greenhouse gas calculators relevant to developing carbon markets in Iowa.

Tool	Scope of analysis	Modeling approach	Ease of use	Level input details required	Geographic scope	Presence of a spatial interface
CaRPE Tool	Whole farm	COMET data-meta modelling	Moderate	Moderate	USA	No
COMET-Farm (Paustian et al. 2019)	Whole farm and off-farm	DayCent modeling framework	Moderate	High	USA	Yes
COMET-Planner (Swan et al. 2020)	Whole farm and off-farm	Informed by field data	Moderate	low	USA	Yes
Cool Farm Tool (Svubure et al. 2018)	Per unit product produced	Informed by field estimates and IPCC emission factors	Moderate	high	Global	No
Fieldprint Calculator (Gillum et al. 2016)	Per unit product produced	Data driven modelling approach	Extensive	High	Global	No

8.3.1.1 CaRPE Tool

The American Farmland Trust hosts The Carbon Reduction Potential Evaluation Tool^{TM}, or CaRPE Tool, which is a web-based tool that allows users to visualize and quantify GHG emission reductions that would result from implementing conservation practices. A detailed description of the methodical approach for this tool is provided in Moore et al. (2020). It is essentially an enhancement of the COMET-Planner, providing land use data layers from the 2017 Census of Agriculture. The CaRPE Tool is the only carbon estimation tool in the United States that can calculate and estimate the implementation cost of agricultural conservation practices such as no-till, crop rotations, and cover crops. Users can insert their costs for a specific practice or else the tool uses the supplied default cost estimates. The designers of the CaRPE Tool focused on making the users fully understand their output. The tool provides graphical visualization, maps, and tabulated data summarizing each scenario scaled to county, state, regional, and even national levels.

8.2.1.2 COMET-Farm

COMET-Farm is a web-based tool for farmers, managers, and ranchers based on the DayCent model. It can predict future and estimate past carbon and GHG dynamics over various time horizons. It provides a relatively complete GHG assessment of methane, carbon dioxide, and nitrous oxide. Users are allowed to construct multiple conservation scenarios to evaluate the effect of changes in, for example, tillage and fertilizer management. It has been extensively validated for simulating carbon and GHG dynamics (Paustian et al. 2018). To assess the impact of mitigation options, users are required to provide management information for both current and future periods, and the model runs on a 10-year projection for each. The results received by the user are the difference between a 10-year period in the future and a 10-year period for the current management practices (Paustian et al. 2018). In essence, the tool is unique in providing a direct spatial user interface, which eases the selection of the assessment location. COMET-Farm also covers a larger number of land uses, including annual row crops, pastures, orchards, vineyards, rangelands, agroforestry, and forestry.

8.3.1.3 COMET-Planner

COMET-Planner is a tool hosted by the USDA Natural Resource Conservation Service and Colorado State University. A detailed methodical description of this tool can be found in Swan et al. (2020). It provides an easy-to-use means of obtaining generalized estimates of GHG mitigation and/or carbon sequestration benefits from implementing a list of specific conservation practices as a component of conservation planning. It was designed to estimate a generalized impact of conservation practices on GHG emission reductions. Thus, the tool is a generalization of the COMET-Farm techniques and has a lower spatial resolution than the COMET-Farm tool as it is intended for initial planning purposes only.

8.3.1.4 Cool Farm Tool

The Cool Farm Tool is an open-source, web-based, integrated, farm-scale GHG assessment and quantification tool developed by the Cool Farm Alliance. It allows farmers to evaluate the impacts of various management options on their environmental performance over time. The tool has a wide range of emission estimation modules (livestock, fuel and energy, agroforestry, irrigation, fertilizer, crop, and soil biomass emissions) designed to compute GHG emissions worldwide. It thus requires a wide range of information to quantify the GHG in CO_2e. While a user can generate estimates in 15–20 minutes, accurate and robust assessments take between 30–50 minutes because of the higher data requirements. Both GHGs and SOC sequestration are estimated from mixed models. Generally, the tool offers a distinct application of the concept of environmental sustainability on a wider geographic scale. It is one of few global GHG calculators that operate globally and in more than 12 languages, including English, French, and Spanish.

8.3.1.5 FieldPrint Calculator

The FieldPrint Calculator was developed by Field to Market to provide farmers with standardized environmental metrics to compare their operation with others in the state or across larger geographical regions (Gillum et al. 2016). The FieldPrint Calculator serves as an interface through which farm operations data can be recorded, shared, and analyzed (Konefal et al. 2019). Farmers or their advisors can enter their production data directly on Field to Market's website to enable the analysis. The GHG emissions metric in the calculator determines emissions based on energy use, nitrous oxide emissions from soils, methane emissions (from flooded fields), and emissions from residue burning. It is based on a series of algorithms to project GHG emissions per unit of crop production. The calculator also includes a separate soil carbon module that uses the USDA-NRCS Soil Conditioning Index (SCI) tool (NRCS 2021). In addition, as of 2021, the FieldPrint also incorporates the COMET-Planner tool as an optional feature to provide interested users with an estimate of the potential amount of carbon that could be stored in their soils following a practice change. The COMET-Planner tool focuses on providing estimates of the impact of a small list of NRCS conservation practices such as cover crops, residue and tillage management, and nutrient management.

8.3.2 Planning Tools

The use of planning tools in agricultural landscapes can assist farmers and their advisors in identifying areas where conservation practices, such as those described in Table 8.1, can cost effectively enhance the production of environmental outcomes (e.g., carbon reduction and storage, improved water quality, enhanced wildlife habitat, etc.). Similar to calculators, heuristic planning tools are often less intensive to operate and interpret than process-based models and are more accessible to a wide range of audiences, including farmers and their advisors (e.g., Ranjan et al. 2019, Gesch et al. 2020). Planning tools can provide stakeholders with desired information about land management and financial tradeoffs associated with decision making (e.g., Roesch-McNally et al. 2017, Zimmerman et al. 2019) and often offer robust spatially explicit outputs and visualizations. As stakeholders, including private industry and public-private partnerships, continue to expand and grow their initiatives to jointly meet agricultural production goals and environmental goals (e.g., nutrient reduction and enhanced water quality, carbon reduction and removal, etc.), planning tools offer the opportunity to meet diverse planning goals and can be used in concert with other GHG and carbon modeling tools (see above and **Chapter 9**) and measurement, evaluation, and valuation tools discussed in **Chapter 3**.

While there are various planning tools available (e.g., InVEST; Sharp et al. 2020), we will focus on the Agricultural Conservation Planning Framework (ACPF) for specific reasons. First, the ACPF was originally developed by the USDA Agricultural Research Service (ARS) to be a publicly available open-source decision support tool that continually leverages public investment in the acquisition, processing, and availability of high-resolution spatial data. Currently, the ACPF is being advanced within the ACPF National Hub housed at Iowa State University, in collaboration with other land-grant institutions (USDA NRCS Cooperative Agreement NR213A750008C007). The mission of the ACPF National Hub is to be the primary database, technology, and expertise center in the United States for government and allied non-governmental conservation partners regarding water quality decision support. Second, the ACPF was designed for application in agricultural landscapes of the Corn Belt and much of its development has featured case studies in Iowa (e.g., Tomer et al. 2015a, Tomer et al. 2015b, Rundhaug et al. 2018, Zimmerman et al. 2019, Gesch et al. 2020, Bravard et al. (under review.) Third, the ACPF is currently being used by agricultural stakeholders in Iowa (e.g., Iowa Soybean Association, IDNR, IDALS, Iowa

NRCS) and throughout the Corn Belt (e.g., The Nature Conservancy) to enhance agricultural and environmental decision making. In addition, several other federal and state agencies, private industries, and NGO organizations are aware of the ACPF and its applications and have provided funding to support the development of the ACPF and its data and applications.

8.3.2.1 The Agricultural Conservation Planning Framework (ACPF)

The Agricultural Conservation Planning Framework (ACPF) is an innovative decision-support tool that identifies opportunities for best management practice (BMP) placement at in-field, edge-of-field, and riparian locations to enhance environmental benefits (e.g., soil health, water quality) in agricultural landscapes (Tomer et al. 2013; Porter et al. 2018). The ACPF can be harnessed for carbon management as a publicly available planning tool that can support decision making involving multiple agricultural and environmental resources. The ACPF operationalizes agricultural land use planning concepts using publicly available geospatial datasets and an ArcGIS toolbox to facilitate field-scale BMP placement and watershed planning at the Hydrologic Unit Code 12 (HUC 12) sub-watershed scale (typically 15,000–40,000 ac; or 6070–16,187 ha) (Tomer et al. 2013; Porter et al. 2018). Using high-resolution LiDAR-based elevation data (e.g., Fig. 8.1), National Agricultural Statistics Service Cropland Data Layer (NASS CDL) land use data (NASS 2021; e.g., Fig. 8.2), NRCS gSSURGO soils data (e.g., Fig 8.3), field boundary data (e.g., Fig. 8.2), and watershed boundary data, the ACPF enables: (1) hydro-conditioning, terrain processing, and hydro-enforcement of elevation data; (2) delineation of the perennial stream network and catchments in a watershed; (3) identification of fields in a watershed that are most likely to contribute nutrients and sediment to surface water; and (4) site opportunities for in-field and edge-of-field BMPs (Porter et al. 2018). The ACPF can assist users in identifying areas of resource concern (i.e., soil and nutrient loss) and where opportunities for conservation may be most well-suited (e.g., high potential nitrogen loss from tile-drained fields; Fig. 8.4). Presently, the ACPF identifies locations for the following BMPs by evaluating site-specific criteria and suitability: drainage water management, grassed waterways, contour buffer strips, bioreactors, nutrient removal wetlands, water and sediment control basins (WASCOBs), riparian buffers, and saturated riparian buffers (Table 8.3; Porter et al. 2018). Many of these practices are covered in **Chapter 4** of this report.

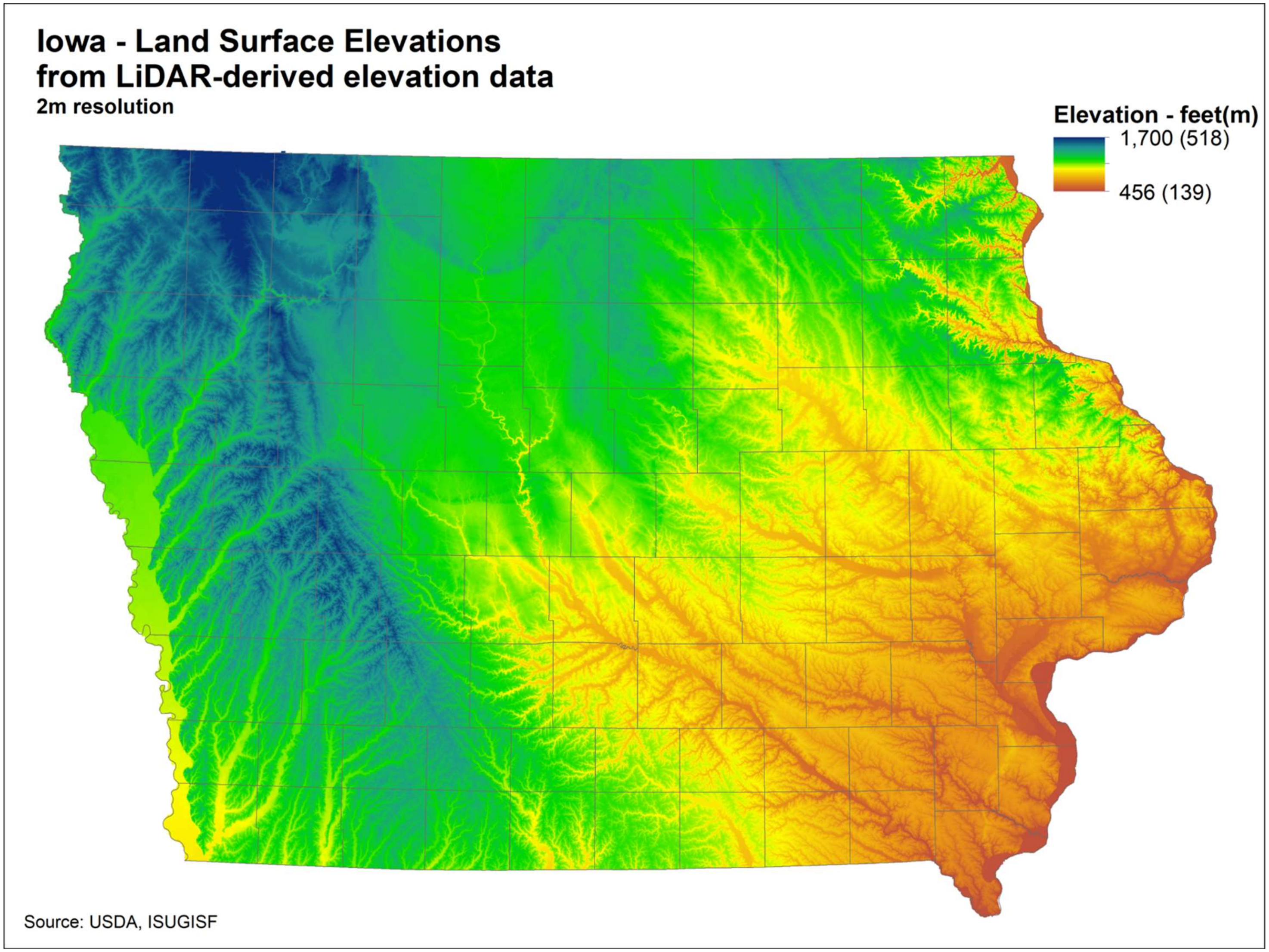

Figure 8.1. Map of Iowa land surface elevation from LiDAR-derived, 2-m resolution elevation data.

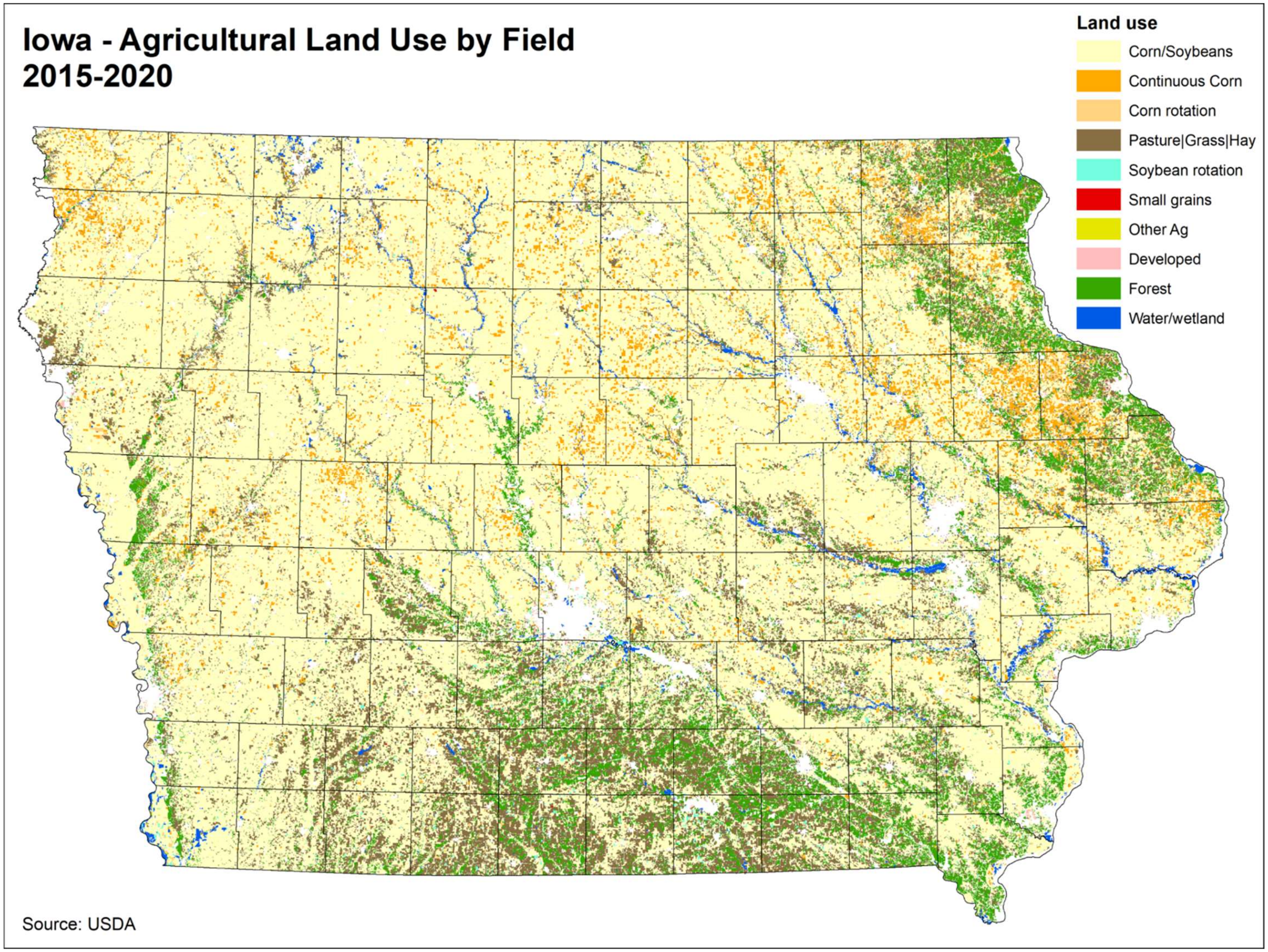

Figure 8.2. *Map of Iowa agricultural land use by field. Because of the number of fields, individual field boundaries are not visible in this map. General land-use categories and rotations created from Tomer et al. 2017 and based on crop rotations.*

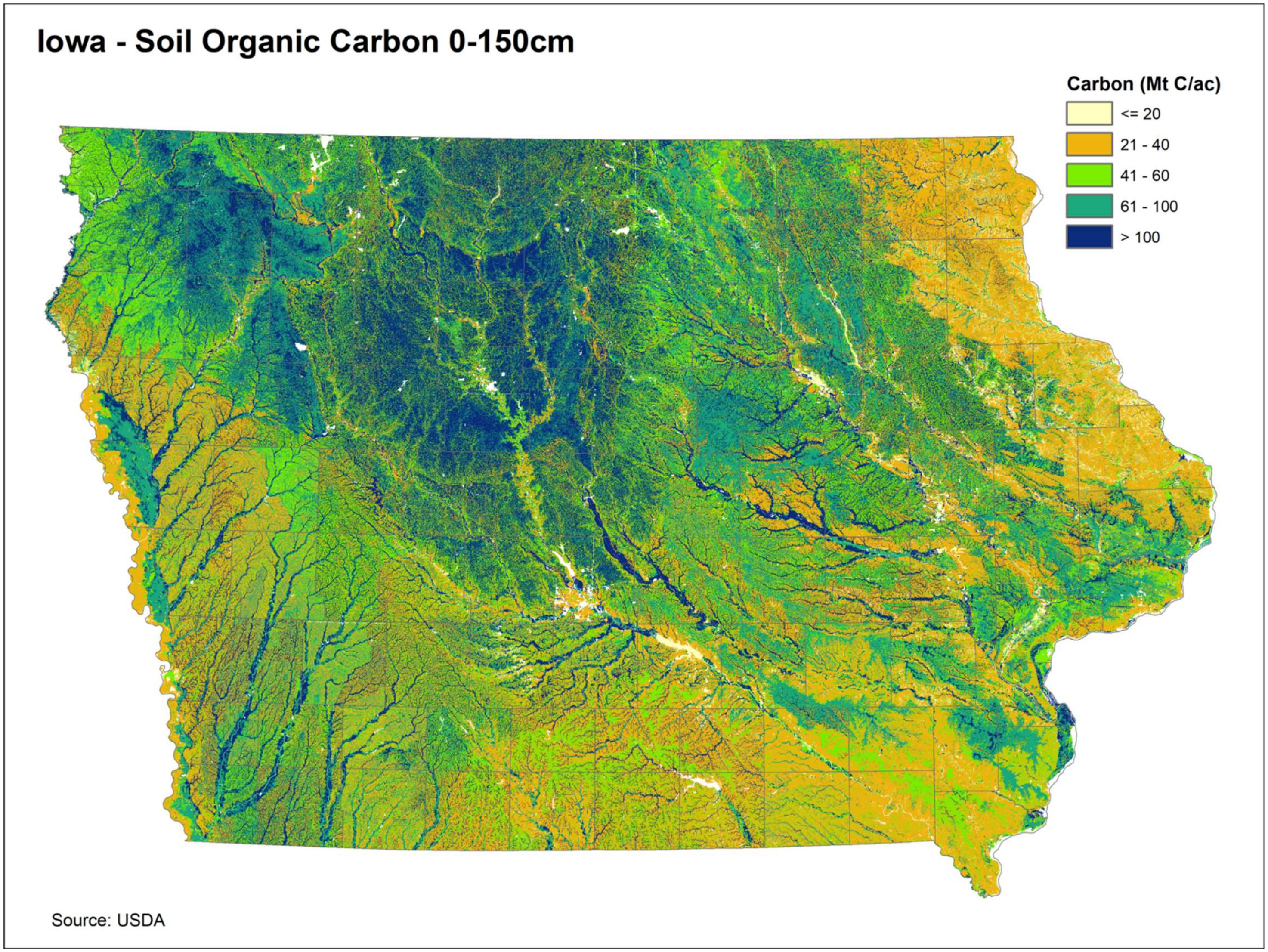

Figure 8.3. Map of Iowa soil organic carbon in 0–150 cm of soil. Data from USDA Soil Survey Geographic (SSURGO) Database.

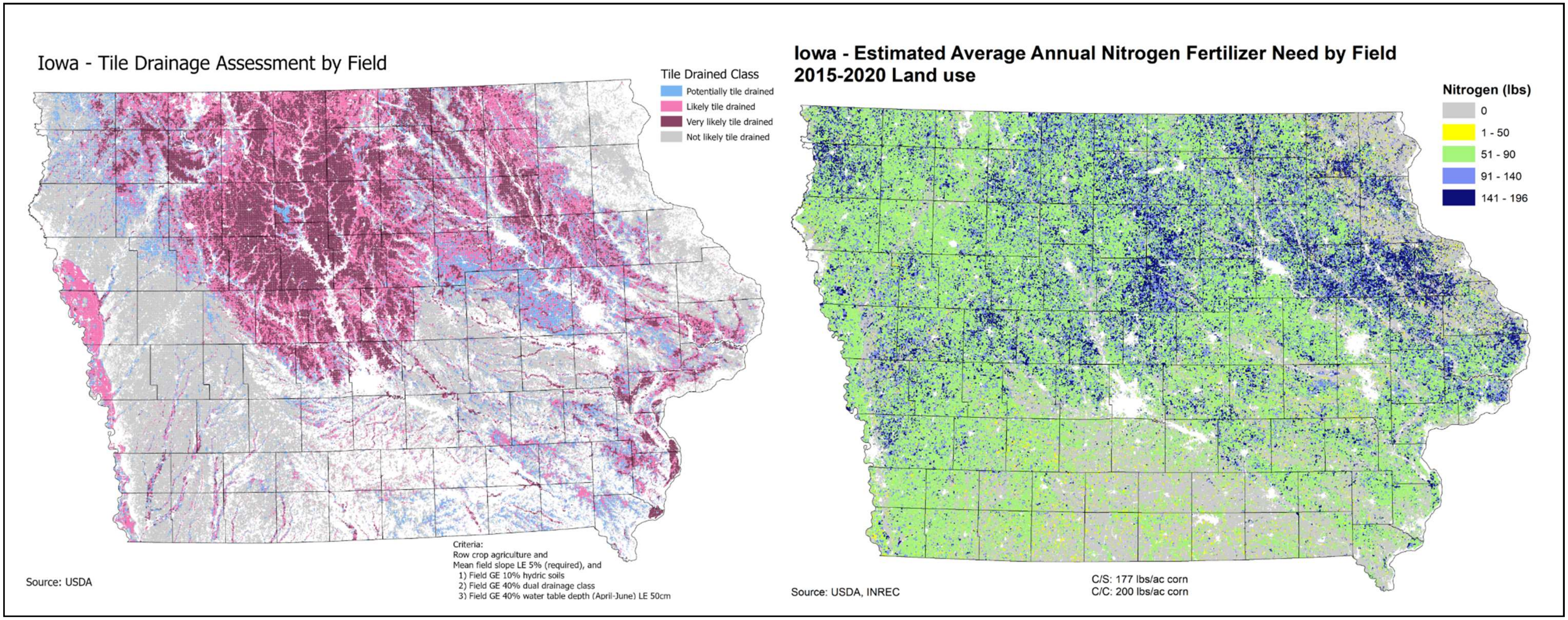

Figure 8.4. Left, Map of Iowa tile-drained classes. Criteria shown in the lower center of the map. Right, Map of Iowa estimated average annual nitrogen fertilized needed by field, based on crop rotations from 2015–2020 land-use data. Nutrient requirements shown in lower center of map. Data may be useful in considering management to mitigate greenhouse gas emissions associated with nitrogen dynamics.

Table 8.3. In-field, edge-of-field, and riparian best management practices (BMPs) and general context for use within the Agricultural Conservation Planning Framework (ACPF) Toolbox. Adapted from Tomer et al. (2013). Other relevant in-field practices can be modeled within the ACPF toolbox (e.g., nitrogen inhibitors, extended rotation, tillage, etc.).

Context	Tile drainage	Surface runoff
In-field	*Where depressions occur:* • Nutrient removal wetlands *Where slopes are least:* • Drainage water management	*Where slopes are steep:* • Contour buffer strips/prairie strips • Conservation cover (e.g., cover crops) • Grass waterways • WASCOBs
Edge-of-field and riparian	*Where surface or tile drainage allow:* • Bioreactor • Nutrient removal wetlands • Farm pond • Saturated buffers	*Where soils are prone to erosion, poor drainage, or are otherwise marginal:* • Perennial cover (cover crops) *Where perennial vegetation does not occur:* • Multi-species forest buffer • Warm season grass buffer/vegetative filter strip

8.3.2.2 The ACPF Financial and Nutrient Reduction Tool (ACPF FiNRT)

The ACPF Financial and Nutrient Reduction Tool (ACPF FiNRT) is an ACPF-compatible, multi-state financial and environmental outcomes tool to quantify costs and environmental outcomes associated with BMPs in conservation planning scenarios (Bravard et al. (under review). Financial and expected field-scale nitrate loss data are used to calculate total long-term costs and cost effectiveness of various conservation plans (Bravard et al. (under review). Financial data were created by calculating direct, long-term annualized costs for BMP installation and management and incorporated enterprise budgets and standard discounted cash flow techniques. BMPs that require removing farmland from production (e.g., nutrient removal wetlands) have opportunity costs associated with their implementation; opportunity costs are spatially determined according to state-relevant, area-weighted-average crop productivity indices and land rent relationships (Zimmerman et al. 2019; Bravard et al. (under review). The tool estimates the nitrogen requirements for each field, based on 6-year land-use data, and evaluates the proportion of nitrogen likely to be lost from the field as nitrate load via leaching informed by Stenbeck et al. (2011) and Lawlor et al. (2008), which summarize nitrate leaching at the Major Land Resource Area (MLRA) scale.

Presently, the ACPF FiNRT quantifies environmental outcomes associated with potential nitrate reduction; sediment and phosphorus will be added to the ACPF FiNRT over the next 1–2 years. The ACPF FiNRT currently quantifies costs and outcomes, when appropriate, for all BMPs included in the ACPF and several additional by-field practices (i.e., cover crops, tillage, living mulch). Other field-level practices of importance for carbon planning, such as those discussed in **Chapter 4**, could be easily integrated into the ACPF FiNRT, including nitrogen fertilizer management and residue management. Current structural practices, such as those detailed in the Iowa BMP Mapping Project, could also be included in analyses to examine previous investment and current outcomes.

Taken cumulatively, the ACPF and ACPF FiNRT distill complex geospatial data, processing, and analyses into outputs for use in conservation planning to meet environmental and financial goals. Outputs from ACPF include attribute tables and maps that identify and visualize relatively high-risk areas of the watershed and potential BMP locations. All data and tools associated with the ACPF and ACPF FiNRT are publicly available and relatively accessible. Results from the ACPF and ACPF FiNRT provide conservation planners, landowners, and land managers with suites of conservation planning options and opportunities (Ranjan et al. 2019; Zimmerman et al. 2019; Bravard et al. (under review).

8.4 Carbon Calculator and Planning Case Studies

To demonstrate the potential utility and application of carbon calculator planning tools, we provide two case studies below. The first case study highlights the application of COMET-Farm and the second case study highlights the application of the ACPF and ACPF FiNRT.

8.4.1 COMET-Farm Case Study

As described above and in **Chapters 2 and 3**, the COMET-Farm tool is widely used in carbon markets to estimate baseline soil conditions and emissions and net sequestration and emissions for proposed future management conditions.

The tool includes a demonstration project example for Iowa State University's Allee Memorial Demonstration Farm near Newall, IA (Buena Vista County). This example illustrates the high impact of nitrous oxide on the estimated CO_2e credits that can be generated and the modest returns available at current market prices for agricultural credits (Fig. 8.5, Table 8.4).

The default values in the demonstration assume a historic management of upland, nonirrigated crops, and no CRP land prior to 1980. From 1980–2000, the default values in the demonstration assume management of nonirrigated, intensive tillage, annual crops in rotation. The baseline management from 2000–18 is a corn-soybean rotation, reduced tillage of cornstalks and intensive tillage of soybean residue, and a spring application of anhydrous ammonia at a rate of 160 lb per acre. The scenario management is the same as the baseline, but tillage is all no-till.

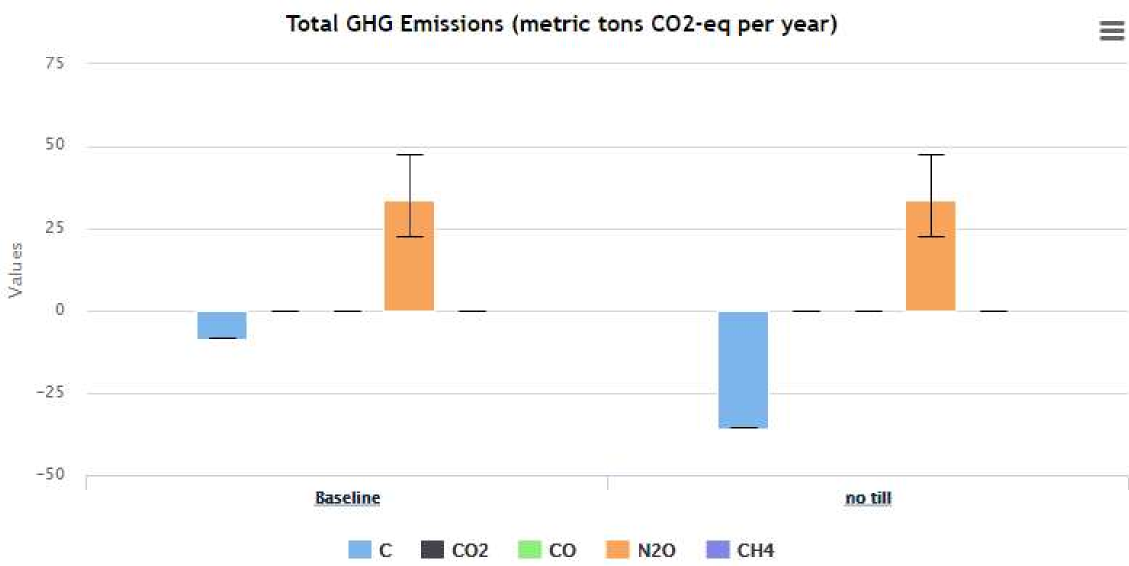

Figure 8.5. COMET-Farm results for transition to no-till system with no change in nitrogen management. GHG = greenhouse gas; C = soil organic carbon; CO2 = carbon dioxide; CO = carbon monoxide; N2O = nitrous oxide; CH4 = methane.

Table 8.4. Example COMET-Farm output. CO_2e = carbon dioxide equivalent.

Category	Subcategory	Baseline	Switch to no-till	Baseline	Switch to no-till
		Mt CO_2e per year		Mt CO_2e per ac per year	
Carbon		-8.6	-35.8	-0.14	-0.60
	Soil	-8.6	-35.8	-0.14	-0.60
Carbon Dioxide		0.0	0.0	0.00	0.00
Nitrous Oxide		33.6	33.6	0.56	0.56
	Direct	26.1	26.1	0.44	0.44
	Soil	26.1	26.1	0.44	0.44
	Indirect	7.5	7.5	0.13	0.13
	Volatilization	3.0	3.0	0.05	0.05
	Leaching and Runoff	4.5	4.5	0.08	0.08
Methane		0.0	0.0	0.00	0.00
Total		25.0	-2.2	0.42	-0.04
Annual Per Acre CO_2e Credits Generated (Mt CO_2e per ac per year)					0.45
Annual Per Acre Potential Credit Revenue at $5 per Mt CO_2e					$2.27
Total CO_2e Credits Generated (Mt CO_2e per year)					27.20
Total Annual Potential Credit Revenue at $5 per Mt CO_2e					$136.00

8.4.2 ACPF and ACPF FiNRT Case Study

To demonstrate the potential applicability of the ACPF and ACPF FiNRT in carbon planning, a case study in the Upper Big Creek watershed in central Iowa is presented here. Upper Big Creek watershed is composed of two, 12-digit hydrologic unit (HUC-12) watersheds. It spans approximately 48,000 ac (19,425 ha) across the Des Moines Lobe. The Des Moines Lobe is a recently glaciated region of Iowa characterized by rich glacial till and relatively low-relief and poorly drained landscapes (Prior 1991). Upper Big Creek watershed terminates in Big Creek Lake (756 ac; 306 ha), which is surrounded by more than 3,459 ac (1,400 ha) of public land, including a state park and a wildlife management area. Over 85% of the watershed is in agricultural production of row-crop corn and soybeans, and cattle production is common in pastures adjacent to streams.

Upper Big Creek watersheds offer an opportunity to demonstrate the application of the ACPF and ACPF FiNRT to jointly address conservation planning for carbon and water quality. Upper Big Creek watersheds have a history of water quality challenges. Big Creek Lake is listed on the USEPA 303(d) List of Impaired Waters for excessive levels of pathogen indicator bacteria *Escherichia coli* (*E. coli*) and algae (IDNR 2020). Listing on the USEPA 303(d) List of Impaired Waters took place for *E. coli* in 2006 and has continued through 2020 (Iowa Department of Natural Resources 2020). A water quality improvement plan, including a total maximum daily load (TMDL), was prepared for the watershed in 2011, specifically with respect to Big Creek Lake (Kiel and Pierce 2011). The majority land use in Upper Big Creek watersheds is associated with row-crop corn and soybeans, which may offer additional opportunities for GHG emissions reduction and carbon removal—and may offer additional income streams (e.g., through carbon markets) to offset BMP costs and meet multiple environmental goals.

Using the ACPF and ACPF FiNRT, we evaluated three different conservation planning scenarios and estimated environmental outcomes (i.e., carbon reduction, nitrate reduction) and costs (Table 8.5, Fig. 8.6). The scenarios included: (1) by-field/in-field practices of cover crops and reduced tillage; (2) edge-of-field practices of contour buffer strips (CBS) and nutrient removal wetlands (NRW); and (3) cover crops, reduced tillage, contour buffer strips, and nutrient removal wetlands.

Estimates of financial remuneration through carbon markets and credits vary widely. Thompson et al. (2021) suggest that carbon prices may vary from $10–$20 per Mt. In our assessment of potential soil carbon contributions, we assume the mean of the estimation in Thompson et al. (2021) at $15 per Mt. Money from carbon markets could be used to offset the cost of practices, such as cover crops, which also have water quality benefits. Similar to the estimation of soil carbon contributions from practices, the value of carbon is variable and may depend on whether the carbon is being purchased as part of a carbon offset program, carbon inset program, or another type of program (e.g., government conservation program). Thus, the ACPF and ACPF FiNRT is agnostic to the value of soil carbon contributions, and stakeholder-driven values for carbon contributions can be easily updated for use in the ACPF and ACPF FiNRT.

Table 8.5. Summaries of three scenarios to demonstrate multidimensional planning for the East and West Big Creek watersheds. Summaries include information from the Agricultural Conservation Planning Framework (ACPF) and ACPF Financial and Nutrient Reduction Tool (ACPF FiNRT). SOC = soil organic carbon; NO_3 = nitrate.

Practice	Acres Removed from Production	Direct Costs	Opportunity Costs	Total Costs	SOC Contributions (Mt)	Potential Market Value for SOC ($15 per Mt)	NO_3 Load Reduced (lb)	Mean $ per lb NO_3 (Range)
Scenario 1: All 45,095 ac (18,249 ha) corn and soybeans in 2 Upper Big Creek HUC-12 watersheds planted in cover crops								
Cover crops	0	$3,288,213	$0	$3,288,213	6,136	$92,046	327,688	$10.05 (8.60-10.30)
No-till	0	--	$0	--	17,387	$260,798	--	--
TOTAL	0	$3,288,213	$0	$3,288,213	23,523	$352,844	327,688	$10.05 (8.60-10.30)
Scenario 2: All ACPF-identified opportunities for nutrient removal wetlands (NRW) and contour buffer strips (CBS) in 2 Upper Big Creek HUC-12 watersheds								
NRW	762	$1,254,244	$148,594	$1,402,838	0	$0	228,105	$6.55 (2.13-22.38)
CBS	256	$7,431	$53,656	$61,087	44	$657	37,033	$2.08 (0.46-6.06)
TOTAL	1,018	$1,261,675	$202,251	$1,463,926	44	$657	265,138	$2.58 (0.46-22.38)
Scenario 3: Combined Scenario 1 (in-/by-field practices) + Scenario 2 (edge-of-field practices)								
Cover crops	0	$3,288,213	$0	$3,283,381	6,136	$92,046	327,688	$10.05 (8.60-10.30)

No-till	0	--	$0	--	17,387	$260,798	--	--
NRW	762	$1,254,244	$148,594	$1,402,838	0	$0	228,105	$6.55 (2.13-22.38)
CBS	256	$7,431	$53,656	$61,087	44	$657	37,033	$2.08 (0.46-6.06)
Total	1,018	$4,545,056	$202,251	$4,747,306	23,567	$353,501	592,793	$7.57 (0.46-22.38)

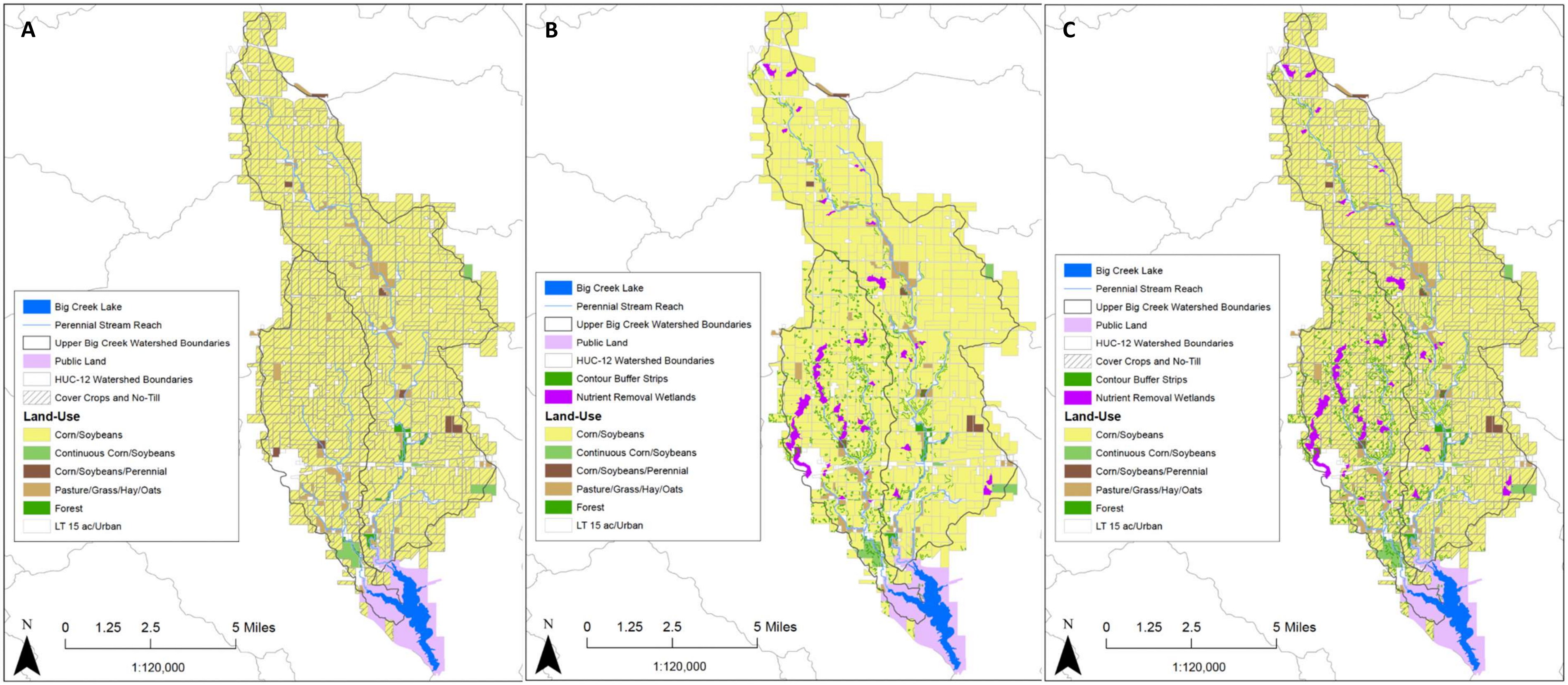

Figure 8.6. (A) Scenario 1: Cover crops and no-till; (B) Scenario 2: Contour buffer strips (CBS) and nutrient removal wetlands (NRW); and (C) Scenario 3: Combined scenarios 1 + 2.

8.4.2.1 Scenario 1: By-field/In-field Cover Crops and No Tillage

In Scenario 1 (Fig. 8.6 A), all 45,095 ac (95% agricultural acres) across 690 fields (92% of agricultural fields) of corn and soybean in Upper Big Creek watershed are planted in a cereal rye cover crop and are not tilled. Costs and nitrate reductions were estimated using the ACPF FiNRT tools. Direct costs associated with cover crop include cereal rye seed, planting, termination, and management. Direct costs in 2021 for cover crops are $64 per ac per year (Bravard et al. (under review). Cover crops do not require opportunity costs because no land is removed from cultivation. No yield decline or increase is calculated for cover crop application. Cover crops are expected to reduce 31% nitrate (IDALS et al. 2017). No cost information is currently available for no-till. No-till currently does not offer a nitrate reduction. In the near future, we anticipate being able to estimate sediment and phosphorus reductions resulting from practices such as no-till and their benefits to surface water quality.

To estimate soil carbon contributions from cover crops, we used coarse estimates from a global meta-analysis (Poeplau and Don 2015) of 0.14 Mt soil carbon per ac per year (0.32 Mt per ha per year). To estimate soil carbon contributions from no-till, we used coarse estimates from **Chapter 4**, which suggest that no-till and very limited tillage systems in Iowa will improve soil carbon stocks at the rate of approximately 0.25 Mt per ac per year (0.62 Mt per ha per year). Here we estimate soil carbon contributions as additive, meaning that for fields with both cover crops and no-till, soil carbon contributions from each practice are added together; research is needed to understand stacked effects. As noted in **Chapter 4**, multiple factors influence a practice's soil carbon contributions; as a planning tool, the ACPF and ACPF FiNRT are intended to provide insights into potential opportunities and offer opportunities for more intensive tools to quantify, monitor, and valuate actual soil carbon contributions. Values for BMP soil carbon contributions can be updated as additional research becomes available.

8.4.2.2 Scenario 2: Edge-of-field Contour Buffer Strips and Nutrient Removal Wetlands

In Scenario 2 (Fig. 8.6 B), all ACPF-identified potential opportunities for contour buffer strips and nutrient removal wetlands were included in the conservation scenario. Contour buffer strips were assumed to be planted as prairie strips with diverse perennial grasses and forbs. Opportunities for contour buffers strips were identified across 303 agricultural fields (40% of agricultural fields) on 256 ac (0.5% of agricultural acres). In the watersheds, 38 opportunities for nutrient removal wetlands were identified. Nutrient removal wetlands and their buffers intersected with 172 fields (23% of fields), covering 762 ac (1.5% of agricultural acres).

Costs and nitrate reductions were estimated using the ACPF FiNRT tools. Direct costs associated with contour buffer strips include land preparation, seed, planting, and management. Direct costs in 2021 for diversely planted contour buffer strips are $29 per ac per year. The nitrate reduction efficiency associated with diversely planted contour buffer strips is 77% (Brittenham 2017). Direct costs associated with nutrient removal wetlands include site planning, design, engineering, and preparation; excavation and soil movement; planting; seed cost; and tile redirection. Direct costs in 2021 for nutrient removal wetlands are $1,646 per ac per year. The nitrate reduction efficiency associated with nutrient removal wetlands is 52% (IDALS et al. 2017). To estimate soil carbon contributions from contour buffer strips, we used coarse estimates from **Chapter 4**, which summarized data from De et al. (2020) for the accumulation of soil carbon since grassland reestablishment across sites in North America. The overall mean rate of soil carbon accumulation was 377 lb per ac per year (0.42 Mt per ha per year). Nutrient removal wetlands are summarized in **Chapter 4**; carbon contributions from nutrient removal wetlands remain unclear. As such, no carbon contributions were included for nutrient removal wetlands. As noted in **Chapter 4**, there is a high level of uncertainty with respect to mean rates of soil carbon accumulation associated with these practices. As data become more available and certainty increases, these values can be updated in the ACPF and ACPF FiNRT to provide information that more effectively reflects BMPs and their outcomes.

8.4.2.3 Scenario 3: Combined By-field/In-field and Edge-of-field Cover Crops, Reduced Tillage, Riparian Buffers, and Nutrient Removal Wetlands

Scenario 3 combines Scenarios 1 and 2 (Fig. 8.6 C) and includes the same costs and nitrate reduction estimations and soil carbon contribution estimations as described above. This scenario is intended to demonstrate combined outcomes from in-field and edge-of-field practices. Conservation practice planning scenarios above highlight potential for stacked practices that produce water quality benefits and contribute to soil carbon outcomes. Addressing water quality outcomes is costly. In the scenarios above, conservation practices range from approximately $1.5 million to $4.7 million (Table 8.5). Participating in innovative private, public-private, and other carbon incentive programs can provide opportunities to offset water quality costs—while simultaneously meeting soil carbon outcomes. For example, in the scenarios above, over $350,00 could be used from carbon efforts to offset water quality costs. Investigation of these dual, win-win practices and their stacked effects will be useful moving forward.

8.5 Research Needs

While tools and calculators are crucial to improving carbon science and supporting evolving carbon markets, models and calculators are only as good as the data used to create them. Currently, data from Iowa relevant to supporting robust predictions of carbon and nitrogen dynamics are limited in spatial and temporal resolution. High-resolution geospatial data, specific to this geographic region, that more effectively and explicitly describe carbon and nitrogen dynamics (e.g., soil carbon), land and livestock management practices (e.g., residue cover, cover crop management, manure practices, etc.), and other biophysical and social landscape characteristics are needed. High-quality data necessitate field-collected data that have been collected using statistically powerful sampling frameworks, allowing data to be applied over large geographic scales with high spatial resolution. Continued investments in high-quality and rigorous data and tools are essential for a variety of reasons. First, ideally, science-based, data-driven tools and calculators are used by field and lab scientists to help pinpoint knowledge gaps and prioritize the most appropriate and useful experimentation, data collection, and management opportunities. Second, calculators are used to generate data to fill spatial and temporal gaps and generate more complete predictions because it is too cost-prohibitive to collect data in all circumstances. Last, tools and calculators generate data or otherwise inform the continued development of carbon and GHG calculators that are used for widespread decision making.

Publicly available, science-driven field-level carbon management planning tools are needed to effectively support agricultural carbon management. While field-level calculators are available, easy-to-use and publicly accessible field-level planning tools are limited. Expansion of publicly available carbon management planning tools and support are important in engaging farmers and advisors in carbon markets and often spur innovation and interest by private industry.

Finally, integration of data and tools used in planning and in measurement, reporting, and verification (MRV) could enhance the efficiency of agricultural carbon management. Opportunities to develop common field-collected and geospatial datasets, common methods and standards for applications of data in tools and calculators, and alignment of tools and calculators used in planning with those used in MRV is an important area of development. Purposeful and intentional design of planning tools and calculators has the potential to reduce transaction costs in the marketplace, more effectively quantifying conditionality, and pursuing carbon management across relevant scales, from the field- to landscape-level.

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Chapter 9. Process-based Agroecosystem Models for Predicting Carbon and Greenhouse Gas Dynamics

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9.1 Highlights

- Process-based agroecosystem models support carbon markets by improving scientific understanding of carbon and greenhouse gas (GHG) dynamics and providing inputs to calculators.
- We describe eight models that have undergone substantial development and have been applied in the Corn Belt.
- We conducted a modeling study using DLEM—the Dynamic Land Ecosystem Model—to illustrate changes in GHG emissions and soil organic carbon (SOC) stocks for the state of Iowa over time.
- Using DLEM, we found a SOC deficit of 700 million Mt carbon for Iowa today compared to the pre-EuroAmerican settlement period.
- DLEM results suggest that large areas of Iowa have acted as a small carbon dioxide sink in the last decade, in the range of -1.2 to 0 Mt CO_2e per ac per year.
- DLEM results also suggest that the current carbon dioxide sink is more than offset by nitrous oxide emissions across nearly all of Iowa, which range 4.9 to 9.9 Mt CO_2e per ac per year.

9.2 Background

Agroecosystem modeling tools, including process-based models and calculators, provide a mathematical representation of agroecosystems that can support carbon markets by accounting for SOC and greenhouse gas (GHG) dynamics. Process-based models are primarily constructed as scientific endeavors to understand complex agroecosystem dynamics, and well-developed models are widely published in peer-review scientific journals. They also may be used to inform decision making and as a platform from which to build calculators for credit estimation and decision making (see **Section 3.4.5** on modeled data and **Section 8.2.1** on calculators). Calculators, by comparison, are modeling tools that are more accessible to a wide range of audiences (e.g., environmentalists, farmers, policymakers, planners, students, students, the general public), created to educate, evaluate, and support decision making.

Process-based models provide mathematical representations of one or more biophysical processes that govern the stocks and flows of energy and materials within an agroecosystem. Different models vary in their treatment of carbon and nitrogen pools and their capacity to model various management practices (Fitton et al. 2019). Differences in their conceptualization of carbon pools could account for differences in their predictions about SOC or GHG dynamics. The availability of multiple models allows users to select a model that best fits their questions or context. However, process-based models are highly complex and often require years of training to properly run and interpret its outputs. At present, process-based agroecosystem modeling is an active area of research in which existing models are being improved and new models are becoming available.

Because in-field measurement is labor and time intensive and costly, carbon programs that buy and sell agricultural carbon credits depend on agroecosystem models to predict changes in SOC stocks and greenhouse gas emissions with changes in farming practices. At present, agroecosystem models are able to provide robust predictions for some measures but not others. For example, models are able to predict maximums and minimums fairly well, but predicting variability is challenging, especially at the field or practice scale needed to account for carbon credits.

In the remainder of this chapter, we (1) provide an overview of the process-based agroecosystem models available to study greenhouse gas and carbon dynamics in agroecosystems, (2) report on a modeling study using the Dynamic Land Ecosystem Model (DLEM) for the state of Iowa to provide an example of the kinds of data needed and the types of outputs process-based agroecosystem models are able to provide, and (3) summarize the research needed to improve model predictions.

9.3 Summary of Existing Process-based Modeling Tools

No comprehensive resource is available that covers and compares all agroecosystem process models, so here we provide brief descriptions of eight widely used process-based models that could be used to improve the basic science on which agricultural carbon markets are built, including GHG and SOC calculators used to estimate carbon credits (for more information, see **Section 3.5.5** on modeled data and **Section 8.3.1** on calculators). Many agroecosystem models are available; those summarized here have undergone substantial development and there has been some effort to apply them to Corn Belt agroecosystems (Table 9.1).

Table 9.1. Summary of process-based models relevant to developing agricultural carbon markets in Iowa. Definitions: SOC = Soil Organic Carbon; GHG = Greenhouse Gas; CH_4 = methane, CO_2 = carbon dioxide; N_2O = nitrous oxide.

Model Name (Key Sources)	Needed Expertise	SOC	GHG Emissions	Spatial Resolution	Timestep	Soil Depth
Agro-IBIS (Kucharick et al. 2013)	Extensive	Yes	CO_2, N_20	Field, state, regional	Hourly	Multiple
APSIM (Holzworth et al. 2014)	Extensive	Yes	CO_2, N_2O*	Field, regional	Daily	Multiple
CENTURY (Parton et al. 1994)	Extensive	Yes	CO_2, N_2O	Field	Monthly	12 in (0.3 m)
Cycles (White et al. 2014)	Extensive	Yes	$C0_2$, N_20	Field, landscape	Daily	Multiple

DLEM (Tian et al., 2010; Lu et al. 2012; Zhang and Tian 2018)	Extensive	Yes	CH_4, CO_2, N_20	Field, regional, global	Daily	39 in (1 m)
DNDC (Gilhespy et al. 2014; Li, 1996)	Extensive	Yes	CH_4, CO_2, N_20	Field, regional	Daily	20 in (0.5 m)
DayCent (Del Grosso et al. 2000; Necpálová et al. 2015)	Extensive	Yes	CH_4, CO_2, N_20	Field	Daily	12 in (0.3 m)
EPIC (Causarano et al. 2008; Kiniry et al. 1995; Williams 1995)	Extensive	Yes	CH_4, CO_2, CH_4	Field, landscape/ watershed	Daily	39 in (1 m)

*APSIM's livestock module represents enteric methane emissions, but oxidation of methane is not represented in the cropping system module.

9.3.1 Agro-IBIS

Agro-IBIS is a detailed, physically based model that simulates carbon and water exchange, soil temperature and moisture at varying depths, and soil biogeochemical processes such as soil nitrogen mineralization and immobilization and dissolved inorganic nitrogen leaching. Agro-IBIS has been evaluated across multiple scales and processes, including photosynthesis from leaf to canopy scale (Vanloocke et al. 2012) and soil biogeochemical processes such as carbon and nitrogen mineralization and immobilization. It has been used to estimate global carbon cycling and was evaluated for its ability to simulate the natural vegetation (forests, grasslands) of the Midwest as well as managed agroecosystems including corn, soybean, wheat, miscanthus, and switchgrass (Bagley et al. 2015; Kucharik et al. 2013) under past, current, and future conditions (Twine and Richter 2013; Walker et al. 2016). Agro-IBIS can be used to simulate changes in carbon storage and respiration at the field, state, and regional scales. Grid cells are 0.3–6.2 mi (0.5–10 km) in size, and the model runs on an hourly time step.

9.3.2 APSIM

APSIM (Agricultural Production Systems Simulator) model is, so far, the most comprehensive process-based model for agroecosystem modeling and research (Holzworth et al. 2018). The model encompasses a broad range of production systems primarily dominated by annual crop production systems alongside other simulated production systems, including horticulture, livestock production systems, forestry, pastoral, and perennial cropping systems (Cichota et al. 2021; Keating et al. 2003). Unlike DNDC (see **Section 9.4.2.6**), APSIM integrates all these production systems into one modeling framework and software package. The model description is broadly described in Holzworth et al. (2014) and Keating et al. (2003). One notable feature of APSIM is its capacity to simulate a wide range of biophysical processes for a variety of crops on a daily time step (Holzworth et al. 2014). The model's central focus is on soil, crops, and water. The soil modeling processes are set up so that the management conditions change the soil's physical and chemical properties, but there is a possibility to reset the soil conditions to the pre-existing conditions for every simulation year. APSIM has a complete crop growth model, which incorporates crop phenology. In this case, the crop includes other perennial components and pasture grasses.

With its modular structure and a very flexible modeling framework, users can alter management strategies by writing new lines of code or using the predefined management nodes loaded from the management library (Holzworth et al. 2018; Holzworth et al. 2014). APSIM has diverse capabilities for modeling different aspects associated with agroecosystems, ranging from simulating crop yields (Puntel et al. 2016), modeling soil and water dynamics, impact of management practices on SOC (Mohanty et al. 2020), nitrous oxide emissions estimation (Thorburn et al. 2010), cover crop biomass yield potential prediction (Marcillo et al. 2019), livestock forage yield modeling (Pembleton et al. 2013), among others. Like in CENTURY (see **Section 9.4.2.3**), soil organic matter is conceptualized into various forms to derive SOC. The residues are components of plant biomass that are not harvested, e.g., plant roots and stover. They are the primary source of soil organic matter and are assumed to be added to the soil through potential decomposition rate, which is driven by residue contact with the soil, soil temperature, and carbon to nitrogen ratio. Everything, including GHG emissions and carbon, are interrelated with the crop growth processes such as carbon capture and nitrogen uptake. For example, nitrogen remaining in the soil after plant uptake is a potential source of nitrous oxide or leachates in the soil. Conversely, the crop biomass accumulation in the form of roots, leaves, and stems creates the soil organic matter, which is the primary carbon source.

Outside the United States, APSIM is relatively well studied with respect to predicting SOC and nitrous oxide compared to the measured stocks. The tool has been well known for its relative representation of the impacts of management activities on SOC (Smith et al. 2019; Mohanty et al. 2020). In the Midwest, emerging research is pinpointing the utility of APSIM to predict crop growth (Archontoulis et al. 2014, 2020; Puntel et al. 2016). Prior studies have also shown that APSIM has the capacity to simulate the effects of biochar on soil organic matter (Archontoulis et al. 2016; Yadav and Malanson 2008). A more recent study by Archontoulis et al. (2020) found that APSIM was able to accurately predict soil nitrogen dynamics. Puntel et al. (2016) also reported that APSIM accurately simulated crop response to mineralization of soil nitrogen. The processes of nitrogen mineralization and uptake are essential for representing SOC and nitrous oxide dynamics. While APSIM can account for carbon and nitrogen processes in the Midwest, more development is needed for it to be able to accurately predict SOC under practices currently being rewarded by carbon markets: no-till, cover crops, and diverse crop rotations.

One notable strength of APSIM is its capability to simulate a wide range of farmers' management decisions, including those that span multiple years, in response to environmental conditions and to predict GHGs and SOC at soil depths exceeding 50 cm. This unique feature gives APSIM a substantial comparative advantage over several process-based models that predict carbon up to 30–50 cm soil depths. These, coupled with its ability to model a diverse range of crop rotations, make the model well-positioned for a long-term assessment of the changes to soil conditions and the environmental sustainability of various farming strategies. Furthermore, the open-access code for APSIM and a large number of users make APSIM suitable for collaboration by scientists from different parts of the world across several disciplines (Cichota et al. 2021).

9.3.3 CENTURY

CENTURY is a soil biochemical model for predicting carbon, nitrogen, and phosphorus levels in the soil. The model underwent major development in the 1990s and was designed for grassland, agricultural croplands, and savanna systems (Parton et al. 1994). CENTURY has been used worldwide to simulate SOC in grasslands, forests, and agroecosystems over monthly time steps, and it is able to simulate for an extended period of time (i.e., centuries). One strength of CENTURY is the fact that it accounts for both soil erosion and deposition, processes that contribute to a significant loss of SOC on a landscape scale. However, integrating both processes while quantifying carbons stocks can be relatively cumbersome. Many studies have evaluated the model performance for the prediction of SOC; e.g., Yadav and Malanson (2008) found a 63% correlation between field-measured and CENTURY's prediction of SOC. CENTURY was used during the compilation of the 1990–2008 *United States Agriculture and Forestry Greenhouse Gas Inventory* (Hohenstein et al. 2011). Many users of CENTURY have shifted to DayCent (see **Section 9.2.5**), a daily time step version of the CENTURY, because of its high temporal resolution.

9.3.4 Cycles

Cycles is a process-based, multi-year, multi-crop, and multi-soil layer model that simulates GHG emissions and changes in SOC in agroecosystems. Cycles can simulate monoculture rotations, double crops, and polycultures. Cycles runs at a daily time step, with hydrology simulated at an adaptive sub-daily time step. The fundamental heat and water transport algorithms are adapted from Campbell (1985) and Stöckle (2008). Cycles includes modules to represent plant growth based on radiation and transpiration use efficiency (Stöckle et al. 2003) coupled soil carbon and nitrogen cycling (White et al. 2014) with phosphorus being currently added, soil water infiltration and redistribution, and the effects of management practices on biogeochemical processes. Model parameterization has already been conducted for the state of Iowa (Armen Kemanian, Penn State University, *personal communication*).

9.3.5 DayCent

DayCent (Daily CENTURY model) is a daily time step version of the CENTURY model used to assess carbon, nitrogen, and water cycling in agroecosystems (Del Grosso et al. 2000). DayCent has been extensively used to predict yield and GHG fluxes in the Great Plains. By shifting from a monthly time step to a daily time step, DayCent was positioned to predict carbon and nitrogen at a very fine temporal resolution, which is important for predicting the highly variable nature of carbon and nitrogen components triggered by short-term environmental events such as rainfall and irrigation. DayCent tracks three soil carbon pools that differ based on turnover rates: active, decadal, and century. DayCent is one of the few models able to simulate a large number of non-carbon dioxide GHGs (Brilli et al. 2017). A case study conducted by Dold et al. (2021) that compared model outputs to field-measured data for a site in Story County, Iowa, revealed that simulated and measured data shared similar trends and magnitudes of change in carbon stocks. This study showed that the soil carbon pool increases in corn years and decreases in soybean years in a corn-soybean rotation under conventional tillage management. Apparently, DayCent is the model that is being used to predict nitrous oxide, carbon, and carbon dioxide in the COMET-Farm tool (Paustian et al. 2018).

9.3.6 DNDC

DNDC, or the Denitrification-Decomposition model, is a biogeochemical process-based model that uses four variable factors—climate (precipitation), management practices (tillage, cover crops), soil factors, and vegetation—to predict carbon and nitrogen dynamics in agroecosystems (Gilhespy et al. 2014; Tang et al. 2006). The model has been developed and deployed worldwide for over 20 years with considerable success. Many validation studies have been conducted in the Midwest (Farahbakhshazad et al. 2008; Jungers et al. 2019; Tang et al. 2006). Besides modeling carbon, DNDC is able to simulate all three major GHGs common in the agroecosystem, i.e., carbon dioxide, methane, and nitrous oxide, (Brilli et al. 2017). It is conceptualized that crop growth is fundamental to the changes in soil biochemical composition and that the uptake of nitrogen and conversion of carbon into biomass can either benefit or negatively affect the soil, depending on the management aspect and type of crop uses. More than 63 cropping systems are now integrated into DNDC. This provides the users with a wide range of crops for investigation, but the users can also create their own cultivar using the supplied crop parameter calculator in the model.

DNDC has a unique modeling advantage over many models associated with alternative modeling practices of nitrogen management, including nitrogen inhibitors, side dressing, slow-release of nitrogen, and urease inhibitors (Zhang and Niu 2016). This is a critical modeling aspect for understanding crop nitrogen dynamics and GHG emissions. The model can also be deployed at a regional scale using geographical information system (GIS) data, which is essential for modeling carbon and GHGs over a larger geographical area (McNunn et al. 2020). Several studies have compared the performance of DNDC for GHG estimation with other models such as DayCent, APSIM, and EPIC, making it one of the most compared models in the peer review domain. The results of these analyses have provided empirical evidence for the utility of DNDC to modeling SOC and greenhouse emissions, in some cases, have triggered improvement. But few of these comparison studies have been done in the Midwest.

However, DNDC is regarded as a one-dimensional model that cannot account for lateral movement of water, unlike its counterparts such as APSIM. In addition, the microscopic organisms in DNDC are assumed to be uniform. In this case, DNDC may not simulate nitrous oxide well following rainfall or irrigation events. In addition, while DNDC includes a wide range of crops, horticultural crops, such as vegetables and fruits that require more intensive management, have consistently received less attention by the modeling community. The frequently modeled crops include corn and barley. There has been limited evaluation studies of the model for horticultural crops.

9.3.7 DLEM

The Dynamic Land Ecosystem Model, or DLEM, is a highly coupled (covering carbon, nitrogen, and water), process-based ecosystem model that simulates plant physiological processes and biogeochemical and hydrological cycles and their variations, driven by natural and anthropogenic forces such as climate variability and change, atmospheric compositions, as well as land use and management practices (Liu et al. 2013; Yu et al. 2018). It is one of a few ecosystem models that has participated in multimodel intercomparison projects for carbon budget assessment (Le Quéré et al. 2018), SOC stock estimation (Tian et al. 2015), and nitrous oxide flux assessment at the global scale (Tian et al. 2019). The geospatial modeling work of the DLEM can operate on a site or any region covering a spatial subset of grid cells at a daily time step. In DLEM, each grid cell is a cohort of multiple plant functional types (e.g., croplands, forests, grasslands, wetlands, etc.). The annual area percentages of cropped acres (including crop types, rotation, and cover crops) in DLEM simulations are identified by multiscale land use and cover maps (e.g., National Land Cover Database-NLCD, and USDA NASS Cropland Data Layer-CDL). The DLEM is able to estimate ecosystem productivity, above- and below-ground biomass, soil carbon content, as well as GHG fluxes at a spatial resolution ranging from meters to kilometers.

Among current land ecosystem models, the DLEM is unique in incorporating multiple environmental drivers, grid-to-grid connectivity through river systems, and simultaneous estimation of carbon dioxide, methane, and nitrous oxide fluxes (Liu et al. 2013; Tian et al. 2010). Its agricultural module has been intensively calibrated and validated against measurements from the Long Term Ecological Research, National Ecological Observatory Network, AmeriFlux, USDA crop yield survey, and USGS gauge monitoring, and has been widely used to quantify the contributions of multifactor environmental changes to ecosystem functions (Chen et al. 2012; Tian et al. 2010; Yu et al. 2018). Substantial improvements have recently been made by incorporating key features of well-established crop models (e.g., APSIM, DSSAT, and CERES) into the DLEM to enhance its capability to simulate crop phenology, biomass allocation, and yield formation (Zhang and Tian 2018).

9.3.8 EPIC

The EPIC, or Environmental Policy Integrated Climate, model is a process-based model that can be used to simulate crop production and SOC for 80 crops (Causarano et al. 2008). The EPIC model is complex and includes weather, hydrology, soil erosion, nutrient cycling, soil temperature, crop growth, tillage, and economics (Kiniry et al. 1995). It has been used to provide a reasonably accurate simulation of crop growth, including details, such as leaf area index, for several locations across Canada and North Dakota (Kiniry et al. 1995). Causarano et al. (2008) used the EPIC model to show that continued adoption of conservation tillage in Iowa would likely lead to an increase in SOC in the surface horizon 8 in (0–20 cm), but the model also showed that if the entire soil profile is included, total SOC in Iowa will decrease over time, primarily as a result of soil erosion. They emphasized that additional data on the spatial and temporal variation in SOC were needed to further calibrate and validate the model. EPIC is integrated within the Agricultural Policy/Environmental eXtender (APEX), which is a landscape and watershed model that also tracks water quality measures.

9.4 DLEM Modeling Example

DLEM is driven by time-series geospatial datasets including daily climate conditions (e.g., average, minimum and maximum temperature; precipitation; shortwave radiation; and relative humidity), annual land use and cover change, monthly concentration of atmospheric carbon dioxide, nitrogen deposition, and agricultural management practices (e.g., nitrogen fertilizer use, irrigation, manure nitrogen application, tillage, and tile drainage; Fig. 9.1). These input databases have been developed and fed into the DLEM model to assess how natural drivers and human activities have affected agricultural production, vegetation, soil carbon stock change, and GHG emissions, as well as land-to-aquatic nutrient loadings (an indicator of land contribution to water quality) across the contiguous United States (Lu et al. 2018, 2020; Yu et al. 2019).The basic model simulation is conducted on plant functional types that coexist in each 5-arc min by 5-arc min (~9.2 km by 9.2 km) pixel at a daily time step. Here, all the model input data are developed at the same spatial resolution for driving the model.

9.4.1 DLEM Model Inputs

The daily climate data used to force the model are harmonized from the Climatic Research Unit (CRU) and North America Regional Reanalysis (NARR) dataset (Mitchell and Jones 2005, Mesinger et al. 2006) to cover the period between 1900 and 2019. Atmospheric carbon dioxide was retrieved from IPCC historical carbon dioxide data and published results (Liu et al. 2013, Wei et al. 2014). Annual gridded nitrogen deposition maps were derived from the interpolated monitoring data of the National Atmospheric Deposition Program (NADP, 2021), extended back to 1900 by using North America model-based nitrogen deposition data (Wei et al. 2014, Dentener 2006).

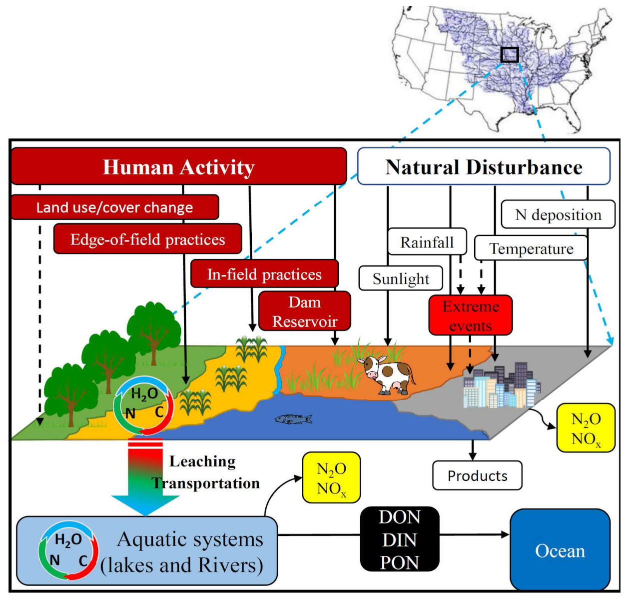

Figure 9.1. Major features of the DLEM include within-grid heterogeneity, grid-to-grid connection, and land-aquatic linkage. Adapted from Lu et al. (2018).

The annual land use/land cover change (LUCC) maps, identifying distribution and percentage of vegetated land (e.g., cropland, grassland, forest, shrub, wetland) and nonvegetated land (e.g., lakes, streams, oceans, glaciers, bare ground, impervious land) in each 5 arc-min pixel were developed by adopting methodologies from Liu et al. (2013), Liu and Tian (2010), and Tian et al. (2010). The annual crop type and rotation maps were reconstructed by using satellite images and the USDA National Agricultural Statistics Service (NASS) survey data (Yu and Lu 2018). In addition to the improved cropland distribution maps, the LUCC database used in this study was developed by including the harmonized forest maps using North American Forest Dynamics (NAFD), Land Use Harmonization data (LUH2) and the reconstructed wetland maps based on state-level wetland area inventory reports (NWI 2021). More details can be found in Lu et al. (2018) and Yu et al. (2018, 2019).

To force the model with long-term management data sets, a series of gridded data is needed to characterize how farmers cultivate land and manage the resources such as water and nutrients. The annual time-series maps of chemical nitrogen fertilizer use (including nitrogen fertilizer use rate, nitrogen fertilizer types, and application timing) were developed across the United States at 1-km by 1-km resolution in a previous study (Cao et al. 2018). In another study (Yu et al. 2020), annual tillage intensity maps since 1998 were reconstructed by combining the farmers' survey data and annual cropland distribution maps. The impacts of crop technology improvement (e.g., plant breeding) on harvest index and crop nitrogen uptake were represented in the recent version of DLEM, and the key parameter values were calibrated against national crop yield records for each crop type obtained from USDA NASS (2021). The irrigation map used was produced by Thenkabail et al. (2009). The tile drainage map of the entire United States was provided by Sugg (2007). Manure nitrogen application data was obtained from Bian et al. (2021).

Figures 9.2 and 9.3 demonstrate the spatial and temporal patterns of selected input drivers that are used to force the DLEM model in this study. The state of Iowa has experienced an air temperature increase at a rate of more than 0.4°C per decade since 1980 (Fig. 9.2a). During the same period, annual precipitation and average shortwave solar radiation increased across large cropland area of Iowa (Fig. 9.2b-d). Compared to other land areas in the contiguous United States, the state of Iowa stands out with high levels of nitrogen input rates, including chemical nitrogen fertilizer uses, manure nitrogen application, and atmospheric nitrogen deposition (Fig. 9.3).

Land-use history data for Iowa indicate substantial expansion in the area of cropland, specifically an increase of 24 million ac (9.9 million ha), from 1850 to 2019 (Fig. 9.4). The land was previously occupied by forests, grasslands, and wetlands. The contemporary land cover map shows that Iowa is now mainly covered by croplands (Fig. 9.5). Major crop types include corn and soybean that are broadly distributed in northern, western, and central Iowa. In northeastern and southcentral Iowa, land cover is dominated by grassland. The 98-ft (30-m) resolution land cover maps were aggregated to a 5 arc-min resolution to reduce the computational cost of model simulations, which inevitably compromised the fine-scale spatial heterogeneity shown in this map.

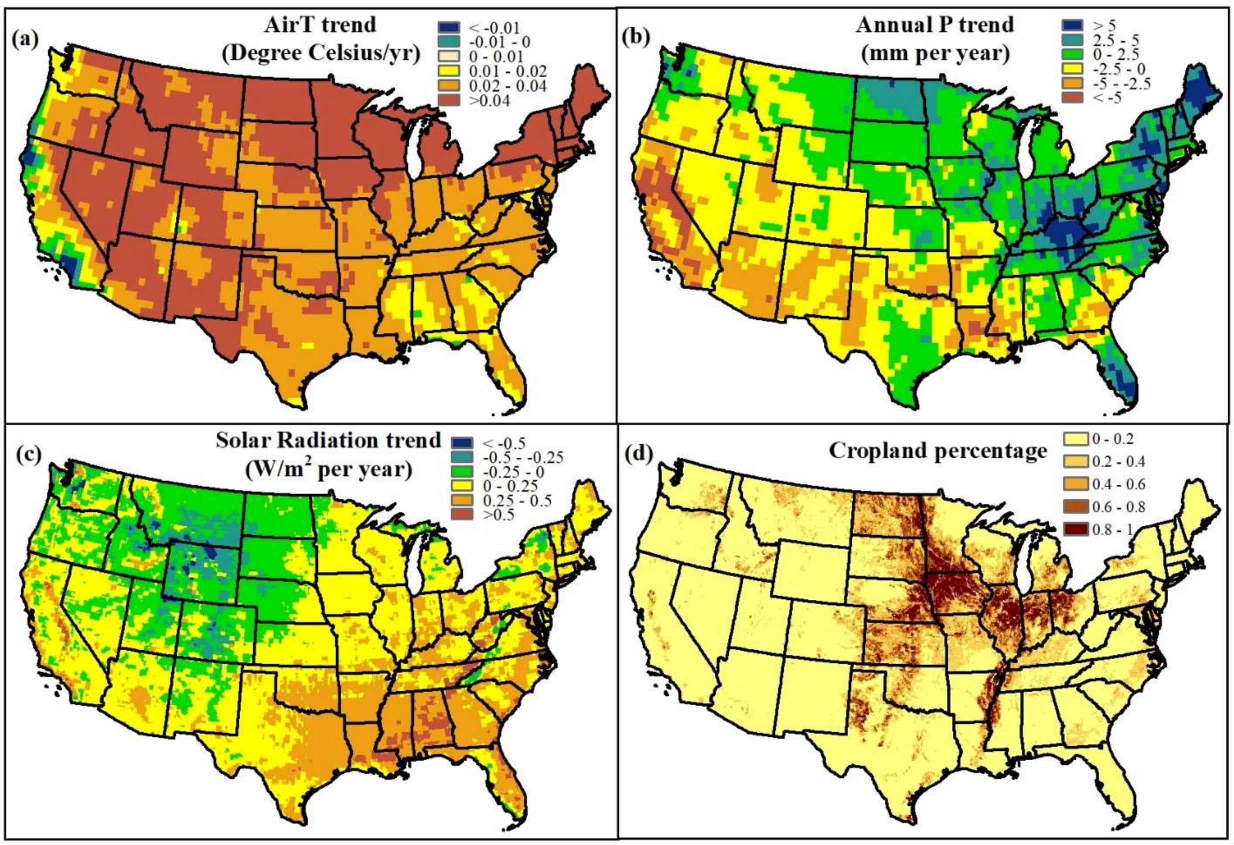

Figure 9.2. Spatial distributions of trends in air temperature (a), annual precipitation (b), and shortwave solar radiation (c) during the 1980–2019 period, and the spatial distribution of cropland percentage (d) in 2019 across the contiguous United States at a spatial resolution of 5 arc-min by 5 arc-min.

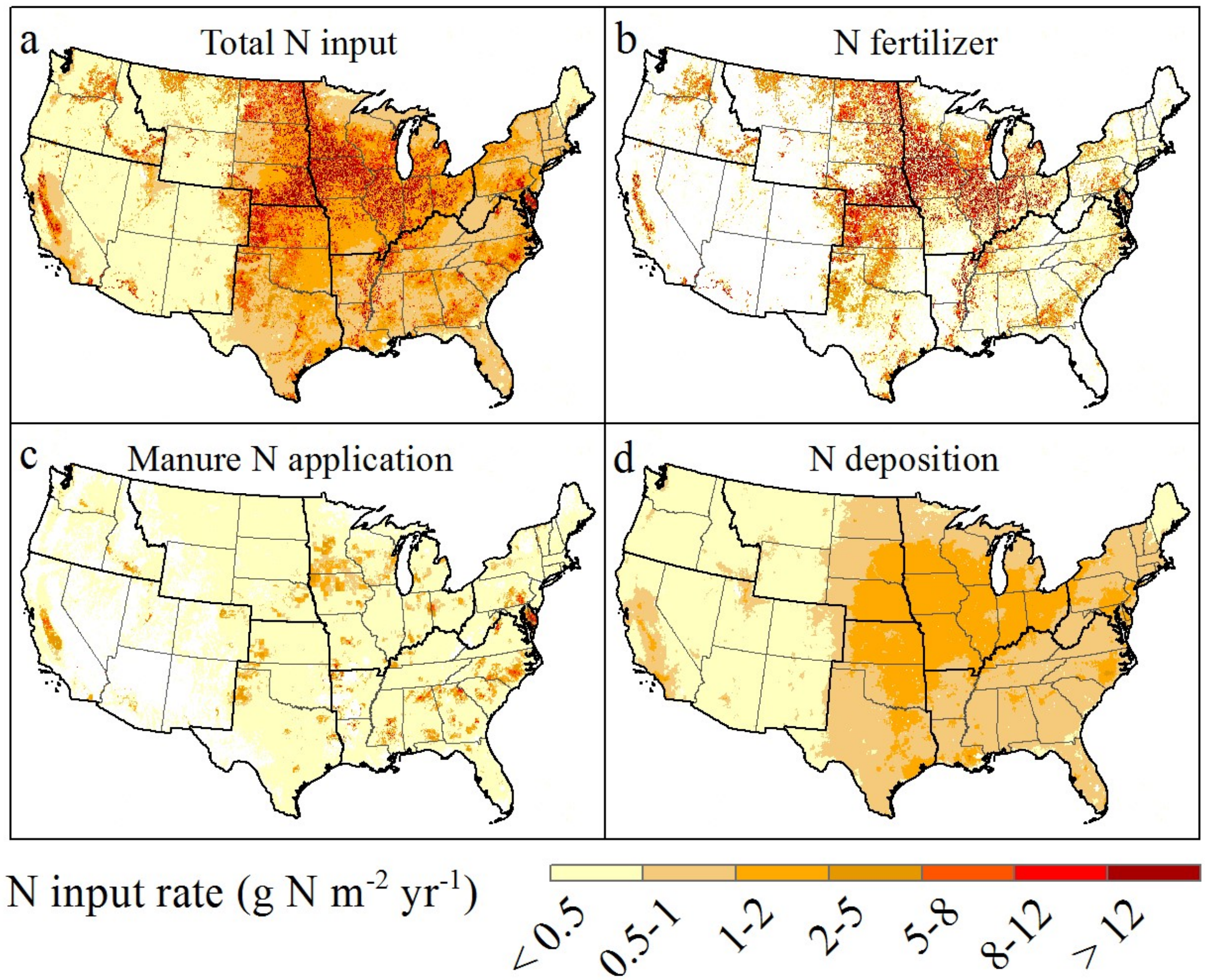

Figure 9.3. Spatial distribution of total nitrogen (N) input (a), nitrogen fertilizer use (b), manure nitrogen *application to croplands (c), and atmospheric* nitrogen *deposition across the contiguous United States in 2019.*

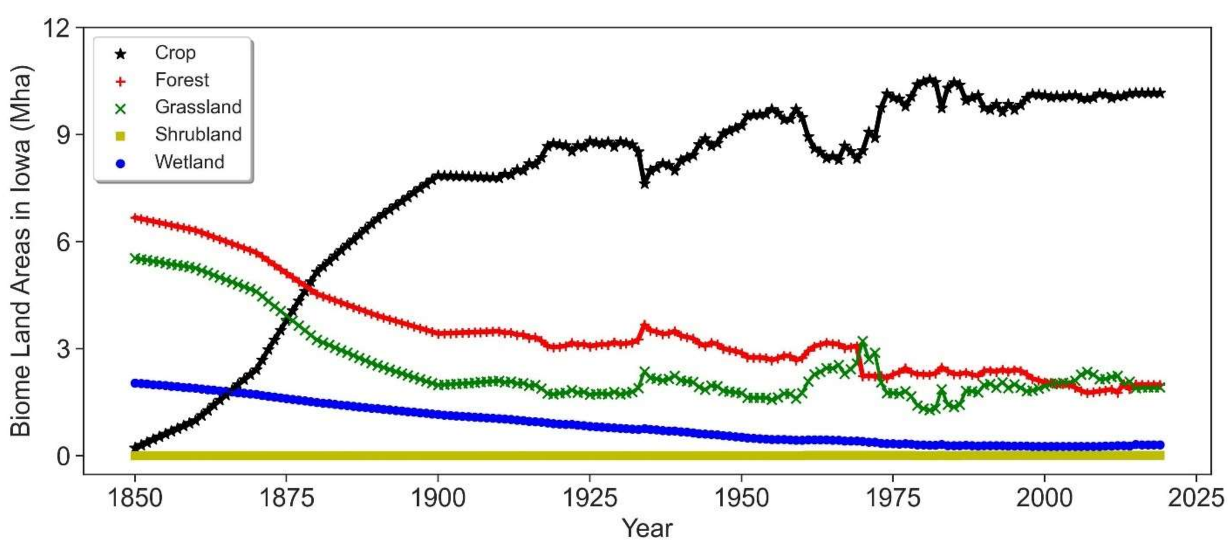

Figure 9.4. Annual time series of land areas occupied by five major biomes in Iowa since 1850.

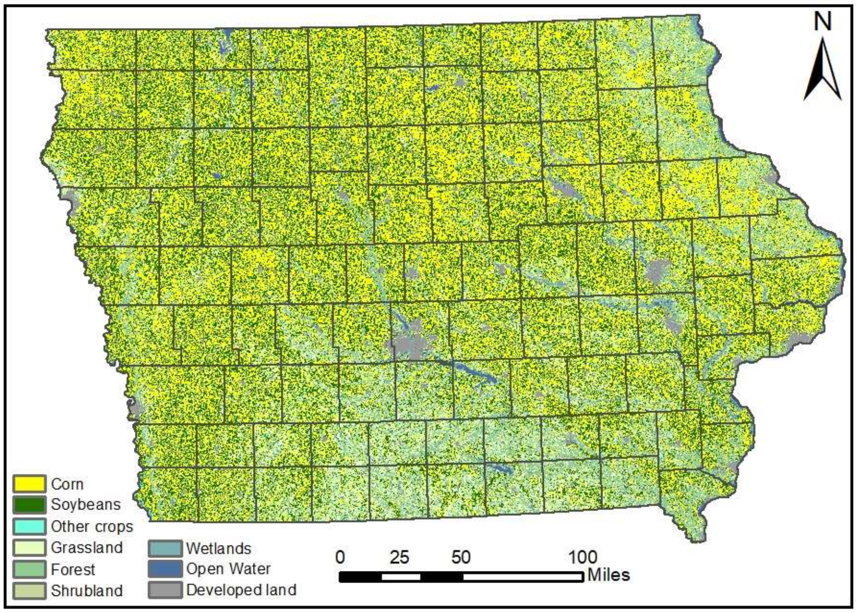

Figure 9.5. Land use and land cover types across Iowa in 2019 (source: USDA CDL-Cropland Data Layer, which served as part of the source data for DLEM simulation in this study and was adjusted based on other United States Census data and resampled to 5-arc-min resolution).

9.4.2 DLEM Model Calibration and Validation

The DLEM model has been rigorously calibrated using observation data obtained from the Long Term Ecological Research (LTER) Network, Long-Term Agroecosystem Research (LTAR) Network, and measurements at flux towers (Liu et al. 2013; Lu et al. 2012; Tian et al. 2010; Yu et al. 2018). Here, 40 and 47 site-year data in the contiguous United States were used to validate model performance in estimating gross primary productivity and SOC content, respectively (Fig. 9.6a-c). In total, data from 116 site-year measurements were used for validating model estimates of nitrous oxide emissions from the soil, which cover major cropping systems (e.g., corn, soybean, barley, wheat, alfalfa) and natural vegetation (e.g., forest, grassland, wetland) (Fig. 9.6d-e). The results showed good agreements between model estimations and observations of annual gross primary productivity, nitrous oxide fluxes, and SOC density in multiple sites across the United States.

Model performance was further validated by estimating the daily magnitude of and variations in soil nitrous oxide fluxes with 7-year observation data (from 2005 to 2011) at a corn-soybean rotation site located in central Iowa (42.05˚ N, 93.71˚ W). It showed that the DLEM model captured the interannual and seasonal variations in agricultural soil nitrous oxide emissions well, although some nitrous oxide pulses in the mid-growing season were missed (Fig. 9.7). It indicated that some model features need to be improved to better capture the huge variations in nitrous oxide fluxes that occurred in just a few days, especially after fertilization in spring during corn planting years. The reasons behind the "data-model mismatch" can be multi-fold: (1) soil nitrous oxide emissions are highly variable in time and space, which limits the ability of relatively few *in-situ* measurements to represent field-scale processes (shown by the large red error bars in Fig. 9.7); (2) the ecosystem modeling estimates could be biased by oversimplifying mechanisms responsible for nitrous oxide pluses following fertilizer application, snow melting, and heavy rainfall events (mechanism understanding needs to be improved, too, which needs close collaboration between modeling and field scientists, and better model structure and parametrizations with high-frequency nitrous oxide flux measurements conducted at more representative sites); and (3) the site-level input data that was used to force the model may have missed critical information of weather events, soil properties, and management practices on records that likely have important impacts on soil nitrous oxide fluxes.

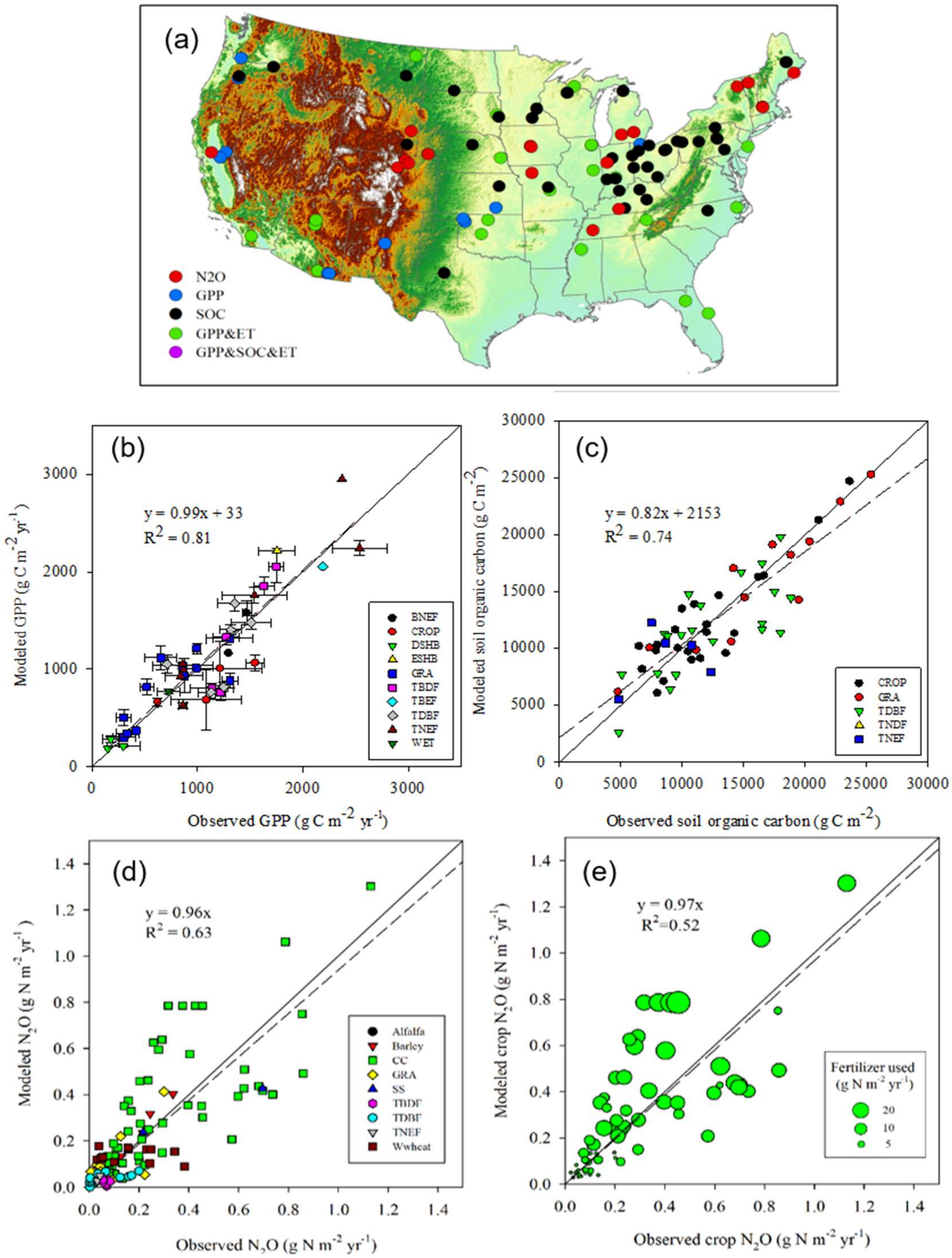

Figure 9.6. Map of model validation sites (a), and comparison between model-estimated and observed annual gross primary productivity (GPP) (b), soil organic carbon density at 1-m depth (c), annual soil N_2O fluxes(d) in major biomes and major cropping systems (e) across the United States error bars in (b) indicate the standard deviation from observed or modeled values in each site. Dash line (in b-e) is the regression line between observed data and modeled results, and the solid line is the 1:1 line. (N_2O: Nitrous oxide; GPP: Gross primary production; SOC: Soil organic carbon; ET: Evapotranspiration; CC: Continuous corn; CROP: Cropland; SS: Soybean; Wwheat: Winter wheat; GRA: Grassland; WET: wetland; TBDF: Temperate deciduous broadleaf forest; TNEF: Temperate evergreen needle-leaf forest)

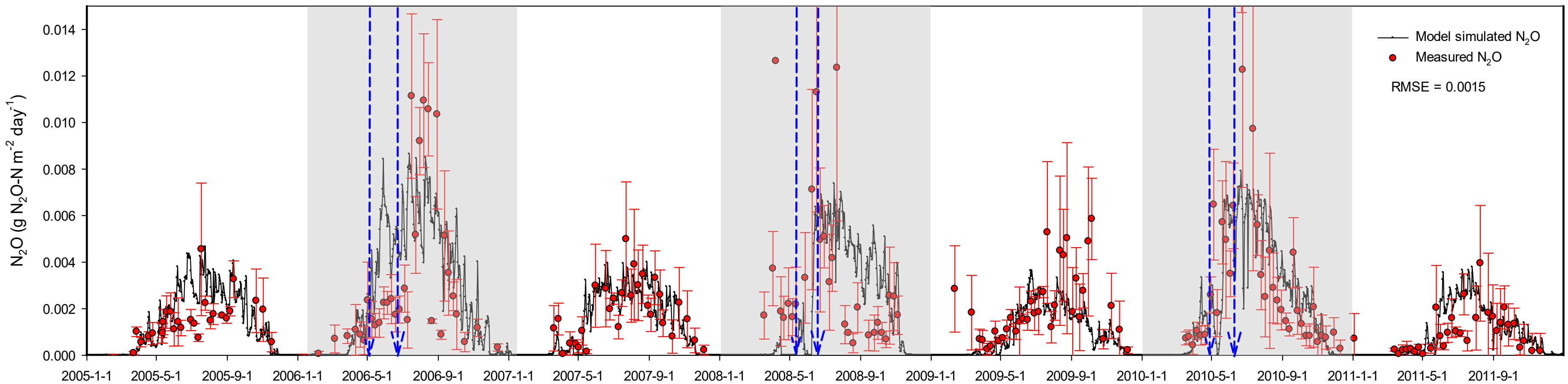

Figure 9.7. Model performance in reproducing the observed daily nitrous oxide (N_2O) emissions at a corn-soybean rotation site in central Iowa (42.05˚ N, 93.71˚ W, Parkin et al. 2016) and personal communication with Parkin. The blue dash arrows indicate two applications of nitrogen fertilizers in corn-planting years). RMSE: root mean square error, an indicator to evaluate the differences between model estimates and nitrous oxide flux measurements at daily time step.

9.4.3 DLEM Modeling Results

From 1850 to 2019, the largest loss of carbon from the vegetation pool was found in forest (300 million Mt carbon), followed by wetland (40 million Mt carbon) and grassland (6 million Mt carbon) (Fig. 9.8a). Carbon in the cropland vegetation pool is close to zero because it was harvested/removed each year and was not counted into the total carbon stock. The total carbon in vegetation across Iowa consistently declined from 530 million Mt in 1850 to 190 million Mt in 2019 at a rate of 2 million Mt carbon per year (R^2=0.83, p<0.001). It is noteworthy that while crop biomass is a large carbon pool in Iowa, it was not included in the carbon accounting because a large amount of above-ground biomass is harvested and potentially consumed elsewhere.

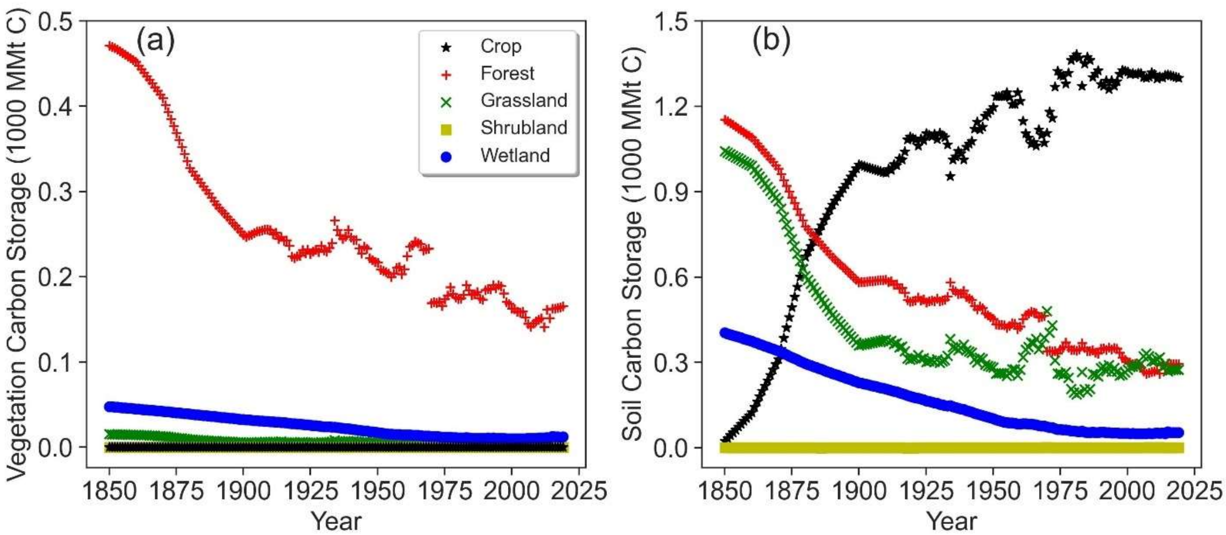

Figure 9.8. Model-estimated carbon storage in vegetation (a) and soil pool (b) of Iowa from 1850 to 2019. Model estimation of soil organic carbon (SOC) stock in this study is for 1-m soil depth. MMt = million metric tons.

With cropland expansion from 0.52 million ac (0.21 million ha) in 1850 to 25 million ac (10 million ha) in 2019 (Fig. 9.4), the model-estimated SOC stock in cropland has increased from 20 million Mt carbon to 1300 million Mt carbon (Fig. 9.8b) in this period. In contrast, other land cover types, including grasslands, wetlands, forests, and shrublands, showed declining SOC stock since 1850, mostly because of land conversion from the non-cropland types to croplands (Fig. 9.4). From 1850 to 2019, the largest SOC loss (860 million Mt) was associated with historically forested areas, followed by former grasslands (770 million Mt) and wetlands (350 million Mt) (Fig. 9.9). Model results show that changes in SOC have resulted in a net loss of 700 million Mt carbon from Iowa soils since 1850 (see the dashed black line in Fig. 9.9).

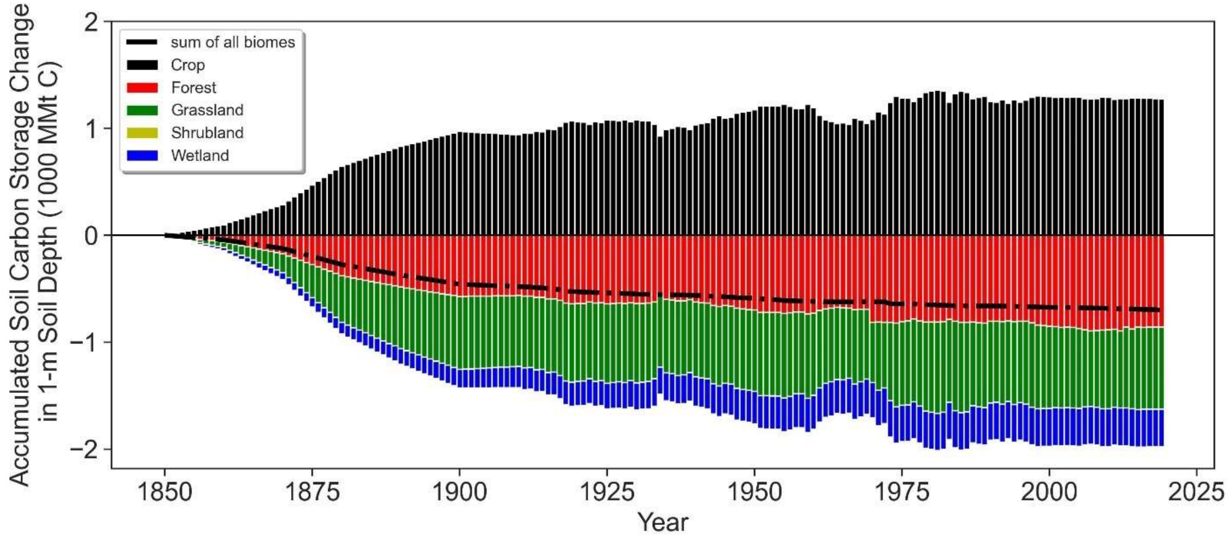

Figure 9.9. Accumulated soil organic carbon (SOC) storage change in major biomes of Iowa from 1-m soil depth since 1850 estimated by DLEM. MMt = million metric tons.

While SOC in Iowa has declined over the past century, the spatial pattern of SOC has become relatively stable since 2000 (Fig. 9.10). Specifically, the greatest mass of SOC per unit area (e.g., >150 Mt carbon per ha) in the year 1900 centered in north-central Iowa, but the spatial extent of carbon-rich soils consistently shrank over time. The density of SOC in Iowa overall demonstrated a statewide decline during the study period. Model simulation shows that the largest SOC density decline (e.g., loss of more than 30 Mt carbon per ha since 1900) occurred in western and central-northern Iowa, primarily driven by cultivation (Fig. 9.10).

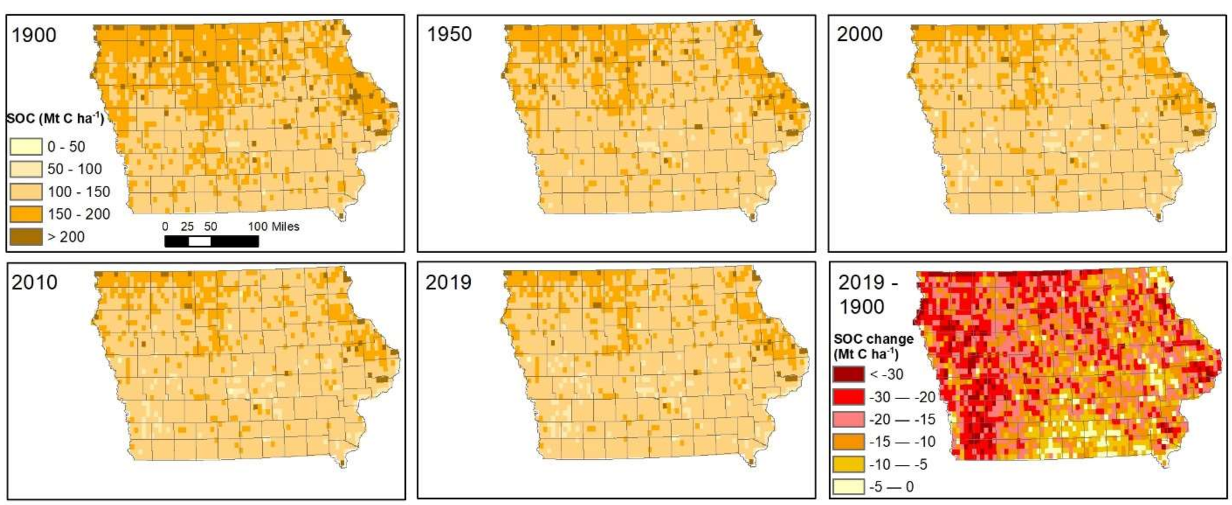

Figure 9.10. Model estimated soil organic carbon (SOC) density in the years of 1900, 1950, 2000, 2010, 2019, and the changes across Iowa during 1900–2019.

SOC stocks decreased in all the Major Land Resources Areas (MLRA) since 1850. The three largest MLRA (103, 104 and 107B) demonstrated the greatest SOC loss of 199, 127, 106 million Mt carbon from 1850 to 2019, which accounted for 29%, 18%, and 16% of the state total soil carbon loss during this period

(Fig. 9.11). The remaining seven MLRA regions together accounted for the other 37% of SOC loss over the study period. Meanwhile, the changes in SOC density (Fig. 9.12) demonstrated a different pattern, with extensive carbon loss in the central and northern Iowa regions. Model simulations indicate a net SOC density decrease of 66.7, 68.2, and 64.3 Mt carbon per ha in MLRA 102C, 107A, and 103, respectively, since 1850. The southcentral and northeastern regions experienced a relatively small decrease in SOC density (e.g., 22.7 Mt carbon per ha in MLRA 109 and 34.3 Mt carbon per ha in MLRA 105) over the same study period. The area percentages of cropland in these two regions are relatively low, and instead grassland dominates the land cover. Low areas of land conversion and land disturbance might result in a low reduction rate of SOC density estimated by the model.

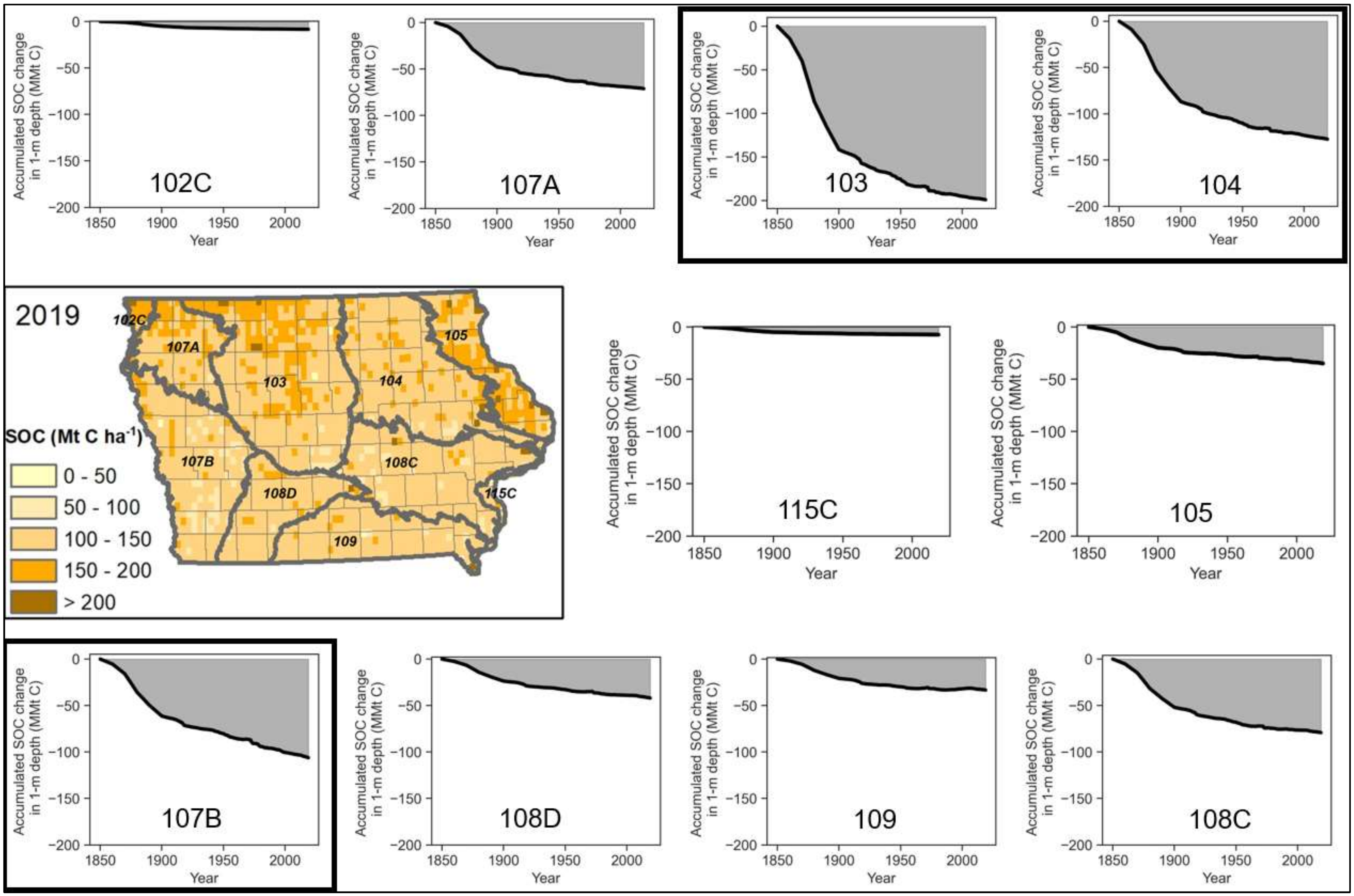

Figure 9.11. Model-estimated cumulative soil organic carbon (SOC) stock change (million Mt carbon, or MMt C) in Major Land Resources Areas (MLRA) regions of Iowa since 1850, with a map showing model-derived SOC density in 2019 and MLRA codes. The MLRA regions with large SOC losses are highlighted.

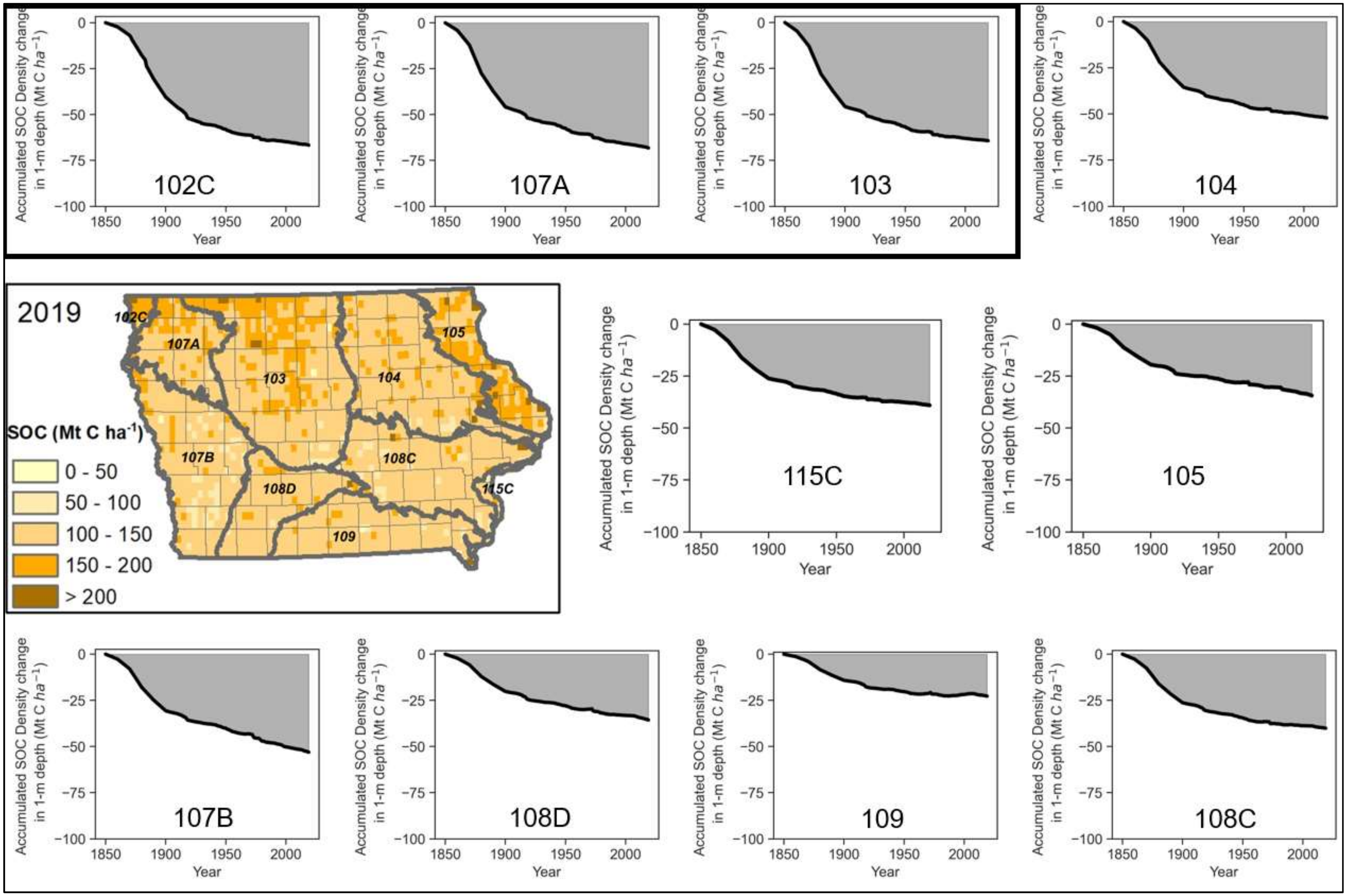

Figure 9.12. Model estimated cumulative changes of soil organic carbon (SOC) density (Mt C per ha) in Major Land Resources Areas (MLRA) regions of Iowa since 1850. The MLRA regions with large declines of SOC density are highlighted.

DLEM was also used to estimate annual fluxes of carbon dioxide, nitrous oxide, and net greenhouse gases (GHGs) in the terrestrial ecosystems of Iowa. The carbon dioxide fluxes were calculated as the year-by-year SOC changes, excluding dissolved and particulate carbon leaching and methane fluxes. Because the carbon dioxide assimilation in crop biomass will be eventually consumed somewhere else, only carbon dioxide emission from the soil pool were counted in this study. Likewise, only soil direct nitrous oxide emissions were included for estimating the net GHG emissions here. The 100-year global warming potential (GWP) was used to convert the fluxes of carbon dioxide and nitrous oxide from gram of carbon and gram of nitrogen into gram carbon dioxide equivalent (CO_2e).

$$F_{CO2^i} = (SOC_{i-1} - SOC_i) - F_{DOC/POC_{leaching}}{}^i - F_{CH}{}^i;$$

$$E_{CO2^i} = F_{CO2^i} \times 12/44;$$

$$E_{N2O^i} = (F_{N2O^i}/28) \times 44 \times 265;$$

$$E_{net^i} = E_{CO2^i} + E_{N2O^i};$$

in which F_{CO2^i} and F_{CO2^i} are fluxes of carbon dioxide and nitrous oxide in the unit of million Mt carbon per year and million Mt nitrogen per year (million metric tons, or 10^{12} g) respectively, and E_{CO2^i} and E_{N2O^i} are emissions of carbon dioxide and nitrous oxide in million MT CO_2e per year. Negative values represent GHG uptake from the atmosphere, while positive values stand for GHG emissions from soils. To calculate CO_2e to assess total GHG balance, 100-year GWPs of 28 and 265 were adopted for methane and nitrous oxide, respectively, according to the fifth IPCC report (Myhre et al. 2013; see **Section 10.2** for more information on calculating GWP). Because soil methane fluxes in Iowa were estimated to be minimal (0.56 million Mt CO_2e per year in the 2010s, less than 2% of nitrous oxide emissions in the same period), we did not include methane into the accounting of net GHG fluxes, but specifically discussed its role in wetlands (see sections below).

The model-estimated GHG fluxes in Iowa demonstrated an increasing trend from 1900 to 2019, despite significant interannual variations found in some years (i.e., 1900, 1918, and 1973). Not surprisingly, the results show that the dominant GHG released from soils in Iowa has shifted from carbon dioxide to nitrous oxide since the 1960s (Fig. 9.13). This implies that the primary human activities contributing to soil GHG emissions in Iowa have changed from cropland cultivation to intensive fertilizer inputs. In the pre-1960 period, before the extensive usage of chemical nitrogen fertilizer, nitrous oxide flux was relatively stable with a min-max range of 3.45 million Mt CO_2e per year in 1917 to 7.92 million Mt CO_2e per year in 1954 (Fig. 9.13). After the 1960s, nitrous oxide became the primary GHG source with an annual increasing trend of 0.5 million Mt CO_2e per year (R^2=0.77, p<0.001). Conversely, carbon dioxide flux after 1960 took a smaller share of net GHG fluxes but showed large inter-annual variations. Overall, the total carbon dioxide fluxes in Iowa shifted from a net source of 9.0 million Mt CO_2e per year in the 1900s to a net sink of -0.11 million Mt CO_2e per year in the 2010s, while nitrous oxide emissions gradually increased from 4.2 million Mt CO_2e per year to 38 million Mt CO_2e per year in the same period (Fig. 9.13).

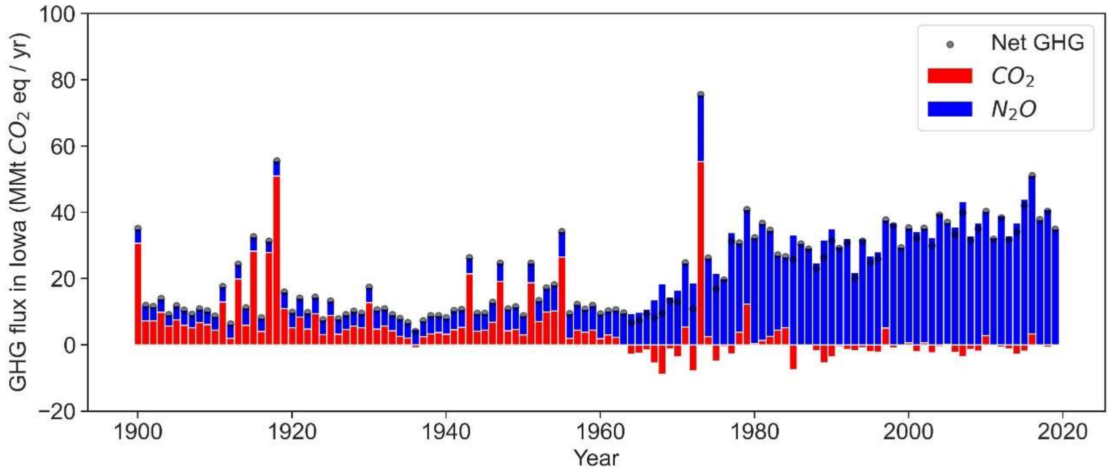

Figure 9.13. Model-estimated annual fluxes of carbon dioxide (CO_2), nitrous oxide (N_2O), and net greenhouse gases (GHG) in the terrestrial ecosystems of Iowa during 1900–2019. Only GHG fluxes from soils are examined here.

The biome-level model results show that GHG fluxes in the 1900s were mainly attributed to carbon dioxide emissions from croplands (6.71 million Mt CO_2e per year, 57% of total CO_2e in 1900s), and the total nitrous oxide emissions from all five major biomes (4.17 million Mt CO_2e per year) accounted for 36% of net GHG fluxes in this decade (Fig. 9.14). In the 2010s, nitrous oxide emissions from croplands were estimated to be the dominant GHG sources, averaged to 36.7 million Mt CO_2e per year. Grasslands and wetlands acted as a carbon sink of, respectively -0.4 and -1.0 million Mt CO_2e per year in the last decade (Fig. 9.14), together offsetting 3.7% of total soil GHG emission in Iowa. Among them, wetlands were the primary source of soil methane emissions (totaled to 0.64 million Mt CO_2e per year in the 2010s), which makes it a negligible net GHG sink (-0.36 million Mt CO_2e per year in the same period). (See also **Section 4.2.10** for additional information on wetlands.)

Although our estimate of net soil GHG fluxes in croplands (i.e., 38.2 million Mt CO_2e per year in the 2010s) is very close to the Iowa Department of Natural Resources (DNR) report for GHG emissions from the agriculture sector (37.82 million Mt CO_2e per year in 2019, IDNR 2020), these two are not an "apples-to-apples" comparison. Iowa DNR GHG report is a "top-down" inventory based on statewide activity data, which includes enteric fermentation, manure management, and agricultural soil management in the accounting of agricultural GHG fluxes. In our modeling estimates, we include soil carbon dioxide and nitrous oxide fluxes driven by human and natural drivers such as changes in climate, atmospheric composition (carbon dioxide concentration and nitrogen deposition), land use and cover changes, crop rotation, fertilizer use, manure management, tillage, and crop technology improvement. Our estimation of nitrous oxide fluxes includes background emission plus its dynamics driven by environmental changes, so it is not directly comparable to the GHG emissions inventory report developed by United States Environmental Protection Agency or Iowa DNR.

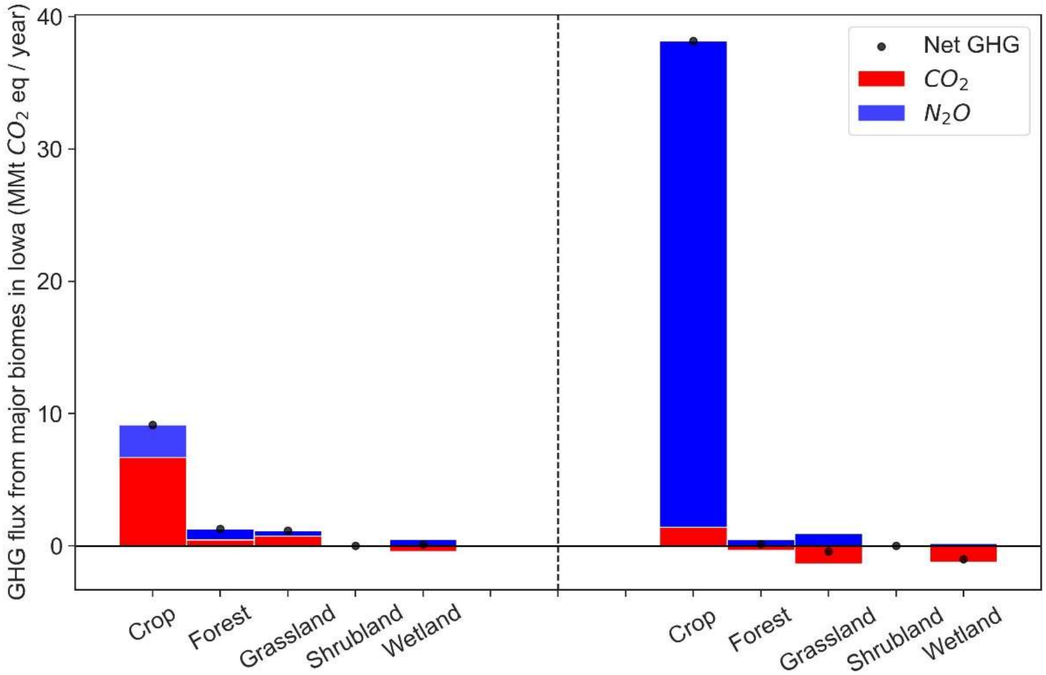

Figure 9.14. Model-estimated decadal average fluxes of carbon dioxide (CO_2), nitrous oxide (N_2O), and net greenhouse gases (GHGs) in five biomes of Iowa during the 1900s (left) and 2010s (right).

The results show that, in the 1900s, the spatial patterns of net GHG fluxes across Iowa were dominated by carbon dioxide fluxes. Specifically, the majority of Iowa was a small net carbon dioxide source (e.g., net GHG flux ranging from 0 to 1 Mt CO_2e per year, Fig. 9.15), while larger net carbon dioxide emissions were restricted to limited areas in central and northern Iowa. During this decade, the spatial pattern of nitrous oxide fluxes was homogeneous across Iowa, which ranged from 0 to 1 Mt CO_2e per year. In the 2010s, large areas within Iowa turned to a small carbon dioxide sink (e.g., carbon dioxide flux falling in the range of -1.2 to 0 Mt CO_2e per ac per year, or -0.5 to 0 Mt CO_2e per ha per year), while the rest of the state remained a net carbon dioxide source (e.g., carbon dioxide flux ranging from 0 to 1 g Mt CO_2e per ha per year). In contrast, nitrous oxide emissions in this decade were significantly elevated in all of Iowa compared to the 1900s. Large areas of Iowa were characterized by high nitrous oxide emissions (e.g., 4.9 to 9.9 Mt CO_2e per ac per year, or 2 to 4 Mt CO_2e per ha per year), except the southcentral part where nitrous oxide fluxes ranged from 2.5 to 4.9 Mt CO_2e per ac per year (1 to 2 Mt CO_2e per ha per year). The model simulation results indicate that the increases in net GHG fluxes across Iowa during 1950–2019 were mainly driven by enhanced nitrous oxide emissions. On the other hand, it implies a robust GHG mitigation potential in agricultural soils.

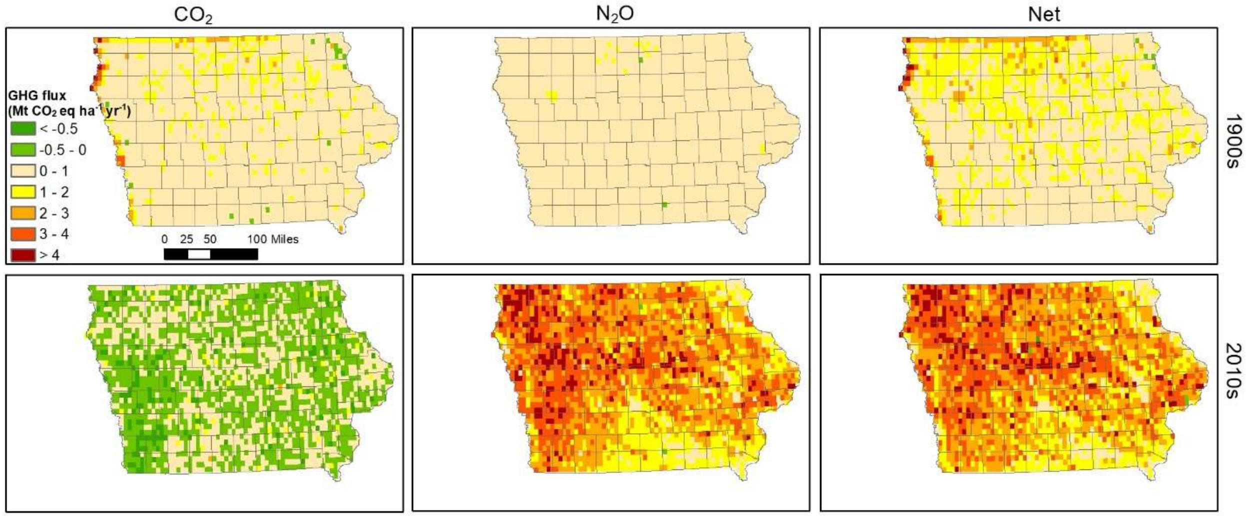

Figure 9.15. Spatial maps of the DLEM-estimated soil carbon dioxide (CO_2), nitrous oxide (N_2O), and net greenhouse gas (GHG) fluxes in the 1900s (upper panel) and the 2010s (lower panel).

9.5 Research Needs

Continued investments in process-based agroecosystem models are essential for a variety of reasons. First, ideally, models are used by field and lab scientists to help pinpoint knowledge gaps and prioritize expensive experimentation and data collection. Second, models are used to generate data to fill spatial and temporal gaps and generate more complete predictions because it is too cost-prohibitive to collect data in all circumstances. Process-based models are needed to support developing carbon markets because they can be used to simulate key outcomes under a range of conditions that cannot be cost-effectively represented with collected data. Last, they generate data or otherwise inform the continued development of carbon and GHG calculators that are used for widespread decision making. They, thus, form the basis of the calculators used to inform investments and management decisions by actors working across sectors, including carbon suppliers and buyers.

Models are only as good as the data used to develop them, however. Currently, data from Iowa relevant to supporting robust predictions of carbon and nitrogen dynamics are limited in spatial and temporal resolution, as is the representation of carbon dynamics at soil depths greater than 12 in (30 cm), with artificial drainage and of novel crop management (e.g., multispecies cover crops, biochar, or digestate additions).

Research that standardizes goals, input data, and assumptions among a set of simulations that are run across more than one model and then compares outcomes, can help elucidate knowledge gaps on model performance and be used to improve the models. Model comparison is an essential aspect of understanding the performance of specific models under certain conditions.

Finally, it is important for developing carbon markets to avoid tradeoffs with food production goals. Thus, agroecosystems models that can concurrently and accurately predict crop yield, GHG emissions, and carbon dynamics are needed. **Appendix 10.2** outlines the types of data and research needed to improve crop yield prediction.

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Chapter 10. Life Cycle Assessment of Carbon Removal Technologies

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10.1 Highlights

- Life cycle assessment (LCA; ISO 14040) is the international standard for comprehensive carbon and greenhouse gas (GHG) assessment in supply chains.
- GREET is a flexible LCA software adaptable to state programs in the United States.
- Iowa's carbon removal potential from engineering technologies ranges from 13 to >35 million Mt carbon dioxide.
- Iowa could develop IA-GREET to guide carbon accounting and removal efforts in the state.

10.2 Background

The growing demand for low-carbon energy has fueled the need for reliable models to track carbon balance across agricultural production supply chains. Government agencies rely on these models to develop policy and validate GHG emissions reduction claims of novel technologies. Most carbon supply chain models follow all or some aspects of the International Organization for Standardization (ISO) standard for life cycle assessment (LCA) (ISO 2006). ISO 14001 defines the general approach for conducting LCA as tracking the environmental impacts of processes, systems, and products.[16] LCA is a broad discipline that has been used in a wide range of applications, ranging from manufacturing to environmental remediation. LCA remains a dynamic field, with scientists developing a wide range of general-purpose and domain specific LCA models. In this section, we describe LCA and review several models that are commonly employed for the assessment of agricultural production supply chains.

The development of LCA dates to the 1960s when concerns over the supply of material and energy resources led to the study of resource availability projections. A standard methodology was not developed until the late 1990s with the establishment of the ISO 14000 series. According to ISO 14040, "LCA addresses the environmental aspects and potential environmental impacts (e.g., use of resources and the environmental consequences of releases) throughout a product's life cycle from raw material acquisition through production, use, end-of-life treatment, recycling and final disposal (i.e., cradle-to-grave)" (International Organization for Standardization 2006). Furthermore, the United States Environmental Protection Agency (EPA) notes that "LCA enables the estimation of the cumulative environmental impacts resulting from all stages in the product life cycle, often including impacts not considered in more traditional analyses (e.g., raw material extraction, material transportation, ultimate product disposal, etc.)" (USEPA 2008). For an agricultural production supply chain, an LCA evaluates the environmental impacts of materials and energy flows associated with feedstock production (cradle),

[16] The registry protocols noted in **Chapter 3** have their own specific requirements for evaluating baseline and project-associated carbon credits in agricultural production that may incorporate elements of LCA sufficient for markets to function, but may not themselves constitute a comprehensive 'cradle-to-grave'/'wells-to-wheel' LCA.

storage, transportation, conversion, consumption, and disposal (grave) as shown in Figure 10.1. This example is for a biomass energy with carbon capture and storage (BECCS) approach.

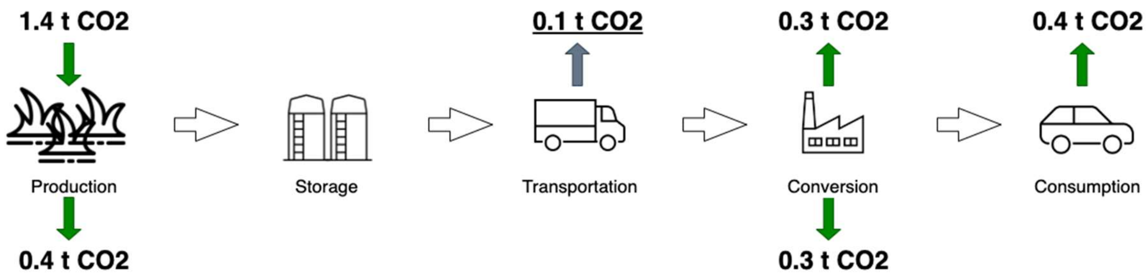

Figure 10.1. Biomass energy with carbon capture and storage (BECCS) and agricultural supply chain carbon flows. Transportation emissions (gray; underlined) include fossil-based fuels. CO2 = carbon dioxide.

The flexibility of ISO 14040 has led to different implementations of LCA models that focus on specific domains. Government agencies in the United States rely on several models to track carbon across agricultural supply chains. The most common model employed in the United States is Argonne National Laboratory's Greenhouse Gases, Regulated Emissions, and Energy Use in Technologies Model (GREET). The United States Department of Energy began to support GREET in 1995, and the EPA adopted GREET to assist in its rulemaking in 2003–04 (Wang 2008). GREET is a model for conducting LCA of fuel and vehicle lifecycles. It has been adopted by several states for state-level rulemaking. The California Air Resources Board (CARB) employs a modified version of GREET (CA-GREET) for evaluating compliance with the California Low Carbon Fuels Standard (LCFS). GREET is a domain specific LCA model for transportation fuels that was recently expanded to include chemicals and other manufactured products.

LCA models depend on lifecycle inventory (LCI) data for the impact factors associated with materials, resources, processes, and systems. Individuals, companies, or institutions collect LCI data from their operations and supply them to LCA databases. There are more than 20 free and commercial databases publicly available for conducting LCA (Di Noi et al. 2017), and the number of LCI sources is growing rapidly. In the absence of LCI data, LCA practitioners rely on models and simulations to estimate impact factors. The EPA, for example, relies on separate models to determine impact factors: (1) Food and Agricultural Policy Research Institute (FAPRI) model for international agriculture, (2) Forest and Agricultural Sector Optimization Model (FASOM) for domestic agriculture, (3) National Renewable Energy Laboratory and United States Department of Agriculture biorefinery models, (4) Oak Ridge National Laboratory crop models, (5) the Global Trade Analysis Project (GTAP), (6) WINROCK models for land use change analysis, and (7) the Energy Information Administration (EIA) National Energy Modeling System (NEMS) for fuel use. EPA models continue to evolve. Policymakers work with different research groups to update the LCI data in GREET and other LCA models. This process remains cumbersome and opaque, but there are growing efforts to facilitate the development of LCA models using open and accessible approaches. This is due to increasing demand from state agencies and private institutions to establish standards that are specific to their sector or region.

The California Air Resources Board (CARB) has established a Low Carbon Fuel Standard (LCFS), which defines Energy Economy Ratios (EER) values for various fuel-vehicle combinations. A fuel pathway carbon intensity (CI) consists of the sum of the GHGs emitted throughout each stage of a fuel's production and use, also known as the "well-to-wheel" or life cycle analysis for the fuel. Figure 10.2

shows the EER-adjusted carbon intensity values of LCFS-certified pathways. California Reformulated Gasoline Blendstock for Oxygenate Blending (CARBOB) has a carbon intensity of 102. All pathways with a lower carbon intensity represent a reduction in carbon emissions relative to gasoline. Pathways with negative carbon intensity scores have net-negative carbon emissions as a result of carbon removal or reduction. Renewable natural gas (RNG) pathways achieve most of the lowest fuel scores (see Bio-CNG and Bio-LNG in Fig. 10.2), exceeding negative 500 g CO_2e per MJ of fuel. These highly negative carbon intensity scores awarded by California's Low Carbon Fuel Standard account for methane from livestock manure being captured and converted into renewable natural gas as opposed to letting it enter the atmosphere where it has more global warming potential (GWP) than carbon dioxide (see **Appendix 11.2** on GWP).

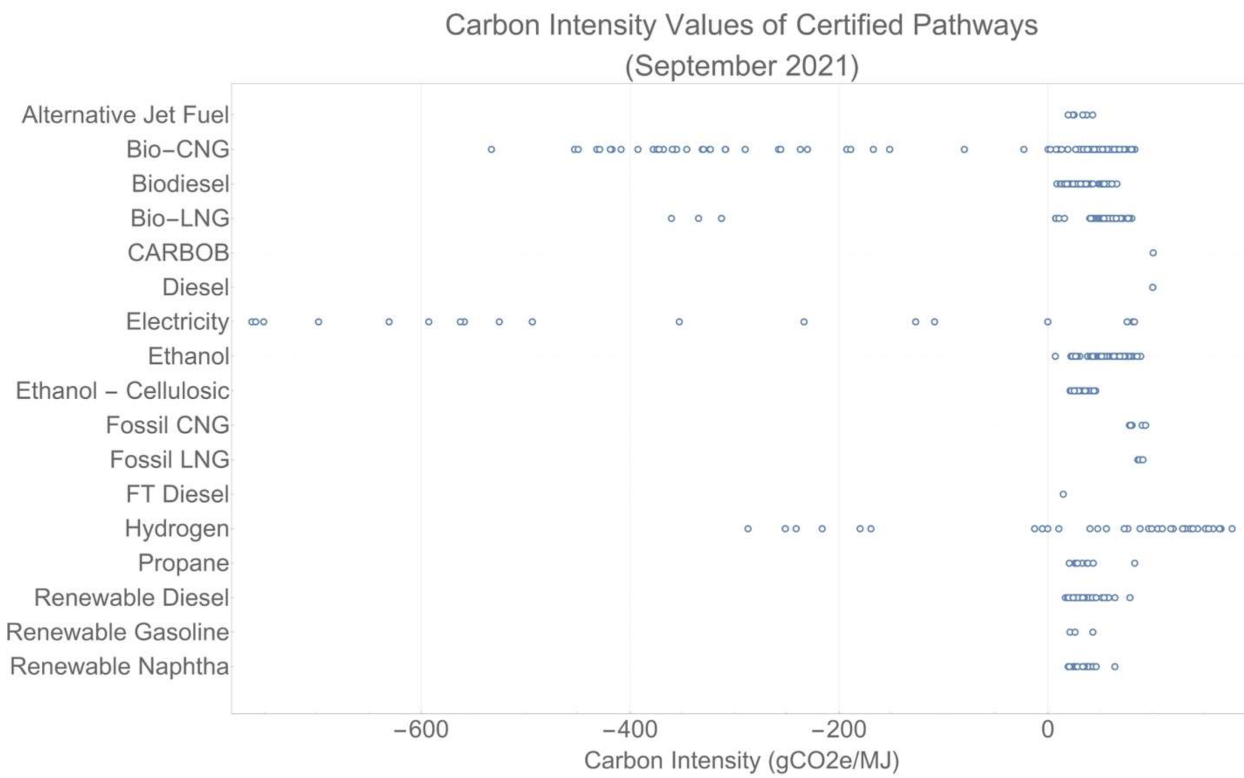

Figure 10.2. Carbon intensity (CI) values of clean fuels pathways approved under California's Low Carbon Fuels Standard; source: California Air Resources Board. Bio-CNG and Bio-LNG are types of renewable natural gas (RNG).

10.3 Carbon Reduction and Removal Potential for Iowa

Iowa has the potential to become a low-carbon or even carbon-negative state through agriculturally based carbon removal. A review of negative emission technologies identified eight low-cost carbon removal technologies (Li and Wright 2020). These technologies convert biomass into power, fuels, and other bio-products while generating biogenic carbon dioxide or carbonaceous solids suitable for carbon sequestration. This analysis focuses on technologies that are commercially available in Iowa or close to technological maturity with the objective of developing a preliminary estimate of the potential to achieve net carbon dioxide removal from the atmosphere.

The eight carbon removal pathway and technology combinations evaluated in this report include:

Biofuel paths with carbon removal and sequestration (CCS):

1. Ethanol from corn grain with carbon dioxide sequestration,
2. Ethanol from corn stover with carbon dioxide sequestration,
3. Diesel from gasification of lignocellulosic biomass with carbon dioxide sequestration,
4. Gasoline from fast pyrolysis of lignocellulosic biomass with biochar sequestration,

Bio-power pathways with CCS:

5. Biomass combustion to power + CCS,
6. Biomass gasification to power + CCS,
7. Anaerobic digestion to power + digestate + CCS, and

Bio-Product Pathway:

8. Biomass fast pyrolysis to produce bio-asphalt with biochar carbon sequestration.

The carbon removal efficiency, cost, and technology readiness levels for these technologies are shown in Table 10.1. Technology readiness level is a metric that measures the maturity of a technology (Mankins 2009). A technology readiness level of 1 is assigned to unproven inventions, and a technology readiness level of 10 refers to commercially ready and mature technologies.

Wright and Li (2020) investigated the carbon removal, or abatement, costs and technology readiness level of negative emission technologies based on a review of the scientific literature (Li and Wright 2020). As shown in Table 10.1, carbon removal technologies have abatement costs that range between \$21 and \$65 per Mt of CO_2e. These costs are competitive with direct air carbon capture and sequestration technologies, which have costs of >\$100 per Mt. These costs are also attractive based on projected carbon prices. However, they are higher than natural solutions, such as reforestation, with costs of about \$11 per Mt (Baker et al. 2020), and they involve some commercialization and sustainability risks.

Table 10.1. Carbon removal technology carbon removal efficiency metrics, abatement costs, and technology readiness levels (adapted from Li and Wright 2020). CCS = Carbon dioxide capture and sequestration. High Technology Readiness Level scores indicate commercially proven technology, whereas low scores correspond to technology in the proof-of-concept and laboratory-stage investigation.

Technology/Pathway	Fraction of Biogenic Carbon Removed (kg C/kg C in Biomass)	Carbon Removal Potential (kg CO_2/kg Biomass)	Carbon Dioxide Removal Cost (\$/Mt CO_2eq)	Technol-ogy Readiness Level
1. Corn Grain Fermentation to Ethanol + CCS	0.20	0.33	\$21.5	10
2. Corn Stover Fermentation to Ethanol + CCS	0.11	0.39	\$50.1	8

3. Biomass Gasification to Diesel + CCS	0.45	0.73	$24.4	7
4. Biomass Fast Pyrolysis of Stover to Fuels + Biochar	0.33	0.53	$32.6	9
5. Biomass Combustion to Power + CCS	0.90	1.47	$64.8	9
6. Biomass Gasification to Power + CCS	0.85	1.39	$50.3	7
7. Biomass Anaerobic Digestion to Power + Digestate + CCS	0.76	1.24	$33.3	10
8. Biomass Fast Pyrolysis of Stover to Asphalt + Biochar	0.60	0.98	$15.0	8

Iowa's carbon removal potential can be estimated based on the resources available, cost of biomass conversion and carbon sequestration, and appropriate socio-economic policy considerations. As an initial estimate, this study gathered data for agricultural residues and ethanol production capacity from the Iowa Biogas Assessment Model (Li and Wright 2020). Iowa produces an estimated 24.4 million Mt of corn stover per year and 4.1 billion gallons of ethanol per year. A carbon removal potential can be determined from these data by adjusting for the performance of each technology.

The carbon abatement graph shown in Figure 10.1 compares the carbon removal potential and abatement costs of select technologies. The estimated carbon removal potentials range between 13.1 and 80.6 million Mt of carbon dioxide. Combusting or gasifying biomass to produce power converts more than 85% of the carbon in the biomass into carbon dioxide that can be captured from a gas stream using carbon capture and sequestration (CCS). These technologies have the highest carbon removal potential but incur the highest costs. Alternatively, CCS can be applied to processes such as ethanol from corn grain, which converts only a portion (~1/3) of biomass into carbon dioxide at a more economical abatement cost. Ethanol from corn grain + CCS is being pursued commercially in the Midwest (Voegele 2021). Asphalt from fast pyrolysis with biochar sequestration has the lowest carbon abatement cost of $15 per Mt owing to its relative technological simplicity and production of bio-asphalt and biochar as carbon sequestration agents. These results suggest that Iowa could invest in a portfolio of technologies to maximize its carbon removal potential. This analysis assumed that each processing technology was not constrained by access to Iowa's feedstock. A more rigorous analysis should address constraints to biomass resources, technology commercialization benefits and risk, and market access based on Iowa's unique characteristics.

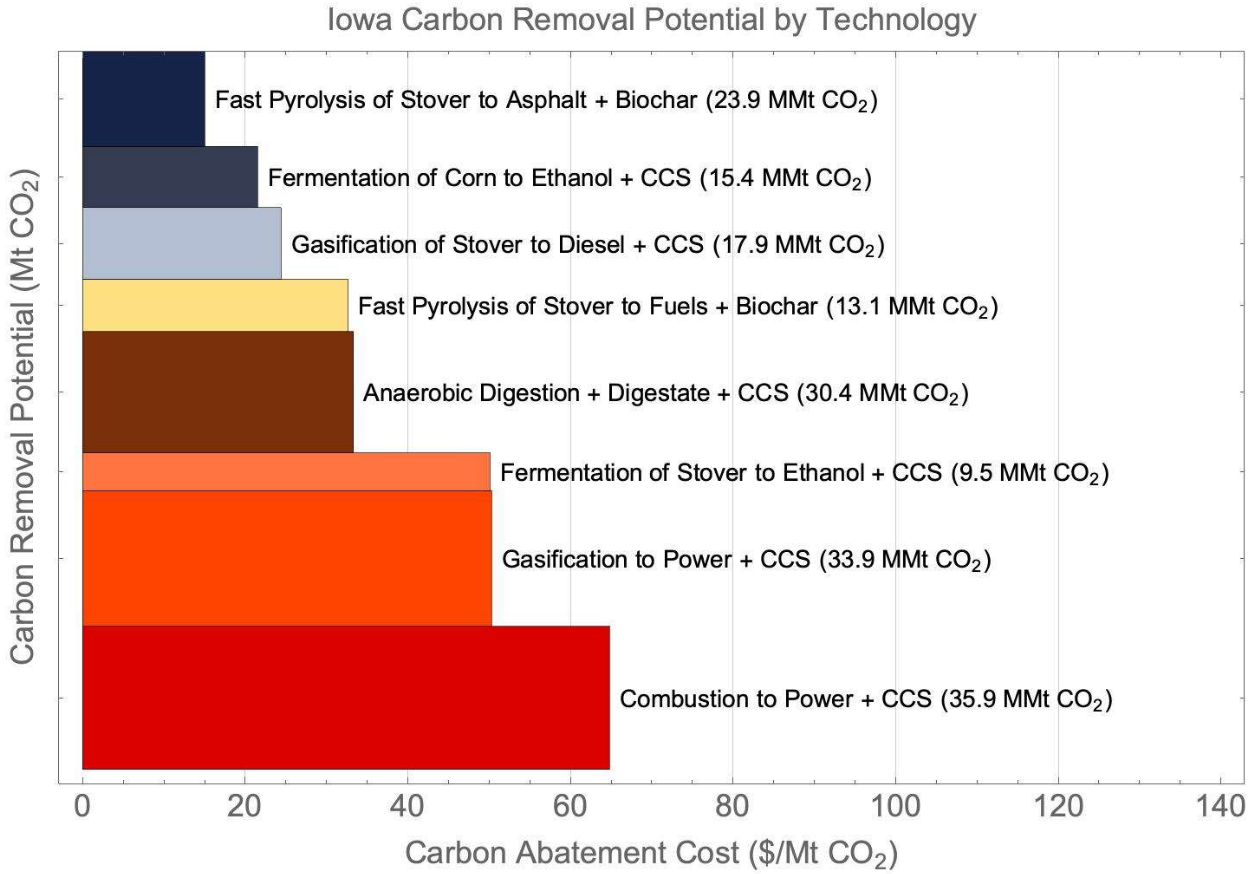

Figure 10.1. Carbon abatement costs of carbon removal technologies converting Iowa's agricultural residues and producing ethanol using carbon capture and sequestration. The carbon removal potential includes overlapping technologies and is not a representation of Iowa's maximum removal potential.

10.4 Research Needs

Iowa's potential to lead in carbon removal rates depends on the synergistic optimization of agricultural practices and carbon removal technologies. LCA studies would help establish carbon removal targets, estimates of the quantities of eligible carbon credits based on the criteria of various carbon markets, and roadmaps for industrialization and commercialization of carbon removal technologies. Iowa features some unique opportunities to leverage its leadership in agricultural production and clean energy systems. Support from the state of Iowa could help establish Iowa-specific LCA databases to guide compliance with incentive programs; incentivize research that will further Iowa's lead in clean bioenergy and bioproduct systems; and establish research centers and public-private institutions that will be supported by future carbon credit revenues.

An example step to address Iowa's LCA research needs in carbon removal is a study by Lawrence Livermore National Laboratory estimating the carbon removal potential of the state of California (Baker et al. 2020). Their study developed strategies for California to reduce emissions from 431 to 0 Mt CO_2 per year by 2045. These strategies were based on seven carbon removal technologies including biochar and carbon dioxide capture technologies; 17 implementation options such as agricultural and livestock practices; and five storage mediums (long-term biomass, soils, geological reservoirs, etc.). These strategies are summarized in Figure 10.2. A similar study would not only align the state of Iowa with the approaches of other states, but also with national and global efforts to establish carbon removal targets.

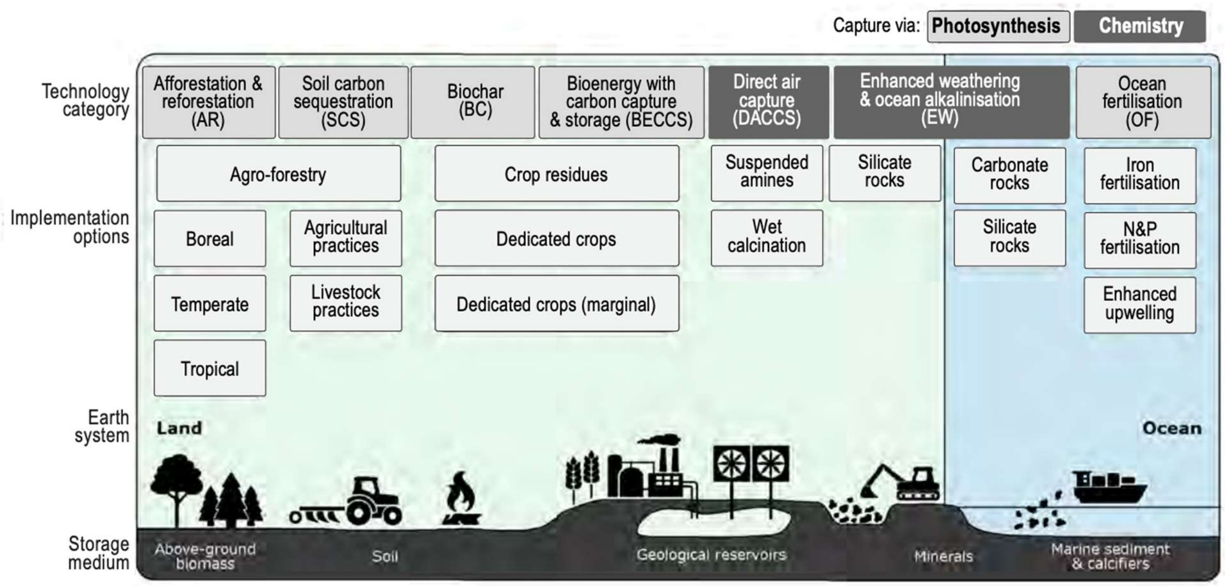

Figure 10.2. California "Getting to Neutral" carbon negative emission technologies (source: Baker et al. 2020).

The state of Iowa could support the efforts of academic institutions, national laboratories, and private organizations to improve lifecycle inventory data that captures Iowa's unique resources and clean agricultural and energy industries. Funding is needed to adapt and incorporate agricultural practices and carbon removal technologies in the GREET database. An Iowa-based GREET database, similar to CARB's database, could be developed in partnership with compliance consulting firms and state institutions. Iowa could encourage collaborations among the Iowa universities to gather emission data, develop climate and earth system models, and create educational materials for students, farmers, and other stakeholders. This support would eventually be supported by revenues from emerging carbon markets, and it could lead to dedicated research and extension centers supported by grants, fees, or industry consortiums invested in this emerging opportunity.

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Appendix 11.1 Glossary of Key Terms

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1. **ADDITIONALITY** – The principle that carbon markets should incentivize activities that would not occur under "business-as-usual" conditions. Carbon markets are structured to motivate new activities not already mandated by regulation or resulting from common business practices.
2. **AGGREGATION (SOIL)** – The binding of soil particles into stable aggregates. Can result from physical, chemical, and biological processes, forming soil aggregates on a range of sizes and stabilities. Micro- and macroaggregates can help protect soil organic matter from decomposition and prolong residence times of soil organic carbon (Jastrow 1996).
3. **AGROECOSYSTEM** – A community of interacting organisms and their environment, but with the intended purpose of agriculture. Agroecosystems have distinct characteristics from natural ecosystems: external inputs of nutrients and energy to enhance productivity of plants or animals, which are under human selection rather than natural selection. Biodiversity tends to be reduced.
4. **AGROECOSYSTEM RESILIENCE** – The capacity of an agroecosystem to resist disturbances (e.g., pests, diseases, droughts, floods) and recover (Liebman and Schulte 2015).
5. **BASELINE** – The carbon level defined as the starting point against which all future measurements will be compared to measure progress.
6. **BASELOAD POWER** – The minimum amount of electric power needed to be supplied to the electric grid at any given time.
7. **BIOCHAR** – The carbonaceous solid produced by the thermochemical conversion of biomass in an oxygen-limited environment (pyrolysis).
8. **BIODIESEL** – A substitute for petroleum-based diesel fuel derived from plant or animal fat-based oils or their transesterified counterparts (US EPA 2021a).
9. **BIOENERGY WITH CARBON CAPTURE AND SEQUESTRATION (BECCS)** – A carbon removal technique that depends on two technologies. Biomass (organic material) is converted into heat, electricity, or liquid or gas fuels (the “bioenergy” step), and the carbon emissions from this bioenergy conversion are captured and stored in geological formations or embedded in long-lasting products (the “carbon capture and storage” step) (American University 2021).
10. **BIOFUELS** – Chemicals derived from biomass that have sufficient volumetric energy densities and combustion characteristics to make them suitable for transportation fuels. They are distinguished from fossil fuels in having lower net greenhouse gas emissions on a lifecycle basis.
11. **BIOGAS** – An energy-rich gas produced by anaerobic decomposition or thermochemical conversion of biomass. Biogas is composed mostly of methane, the same compound as in natural gas, and carbon dioxide. The methane content of raw (untreated) biogas may vary from 40–60%, with carbon dioxide making up most of the remainder with small amounts of water vapor and other gases.

Biogas can be burned directly as a fuel or treated to remove the carbon dioxide and other gases for use just like natural gas. Treated biogas may be called renewable natural gas or biomethane (USEIA 2021).

12. **BIOGENIC CARBON** – Biogenic carbon dioxide emissions from bioenergy are generated during the combustion or decomposition of biologically based material. Biogenic feedstocks differ from fossil fuels in that they may be replenished in a continuous cycle of planting, harvesting, and regrowing. The same plants that provide combustible feedstocks for electricity generation also sequester carbon from the atmosphere (USEPA SAB 2012).
13. **BIOMASS** – Renewable organic material that contains stored chemical energy from the sun. Plants produce biomass through photosynthesis. Biomass can be burned directly for heat or converted to renewable liquid and gaseous fuels through various processes (USEIA 2021).
14. **BIOMASS ENERGY** – The energy from plants and plant-derived materials. Frequently shortened to *bioenergy* (NREL 2021).
15. **CAP AND TRADE** – A scheme for reducing pollution emissions by imposing a ceiling or cap on emissions. Emitters exceeding their allowance can purchase or trade allowances with entities that do not fully utilize their cap.
16. **CARBON BORDER TAX/ADJUSTMENT** – Carbon tariffs against imported products that have a higher carbon footprint than products made within the country establishing the tariff.
17. **CARBON BUDGET** – Refers to three concepts in the literature: (1) an assessment of carbon cycle sources and sinks on a global level through the synthesis of evidence for fossil fuel and cement emissions, land-use change emissions, ocean and land carbon dioxide sinks, and the resulting atmospheric carbon dioxide growth rate. This is referred to as the global carbon budget; (2) the estimated cumulative amount of global carbon dioxide emissions that is estimated to limit global surface temperature to a given level above a reference period, taking into account global surface temperature contributions of other greenhouse gases and climate forcers; (3) the distribution of the carbon budget defined under (2) to the regional, national, or subnational level based on considerations of equity, costs, or efficiency (IPCC 2018).
18. **CARBON CREDIT** – Unit certified by a carbon credit program or standard that can be traded in carbon markets, representing one metric ton of carbon dioxide equivalent.
19. **CARBON CYCLE** – The flow of carbon (in various forms, e.g., as carbon dioxide, carbon in biomass, and carbon dissolved in the ocean as carbonate and bicarbonate) through the atmosphere, hydrosphere, terrestrial and marine biosphere, and lithosphere (IPCC 2018).
20. **CARBON (DIOXIDE) CAPTURE AND SEQUESTRATION (CCS)** – A process in which a relatively pure stream of carbon dioxide from industrial and energy-related sources is separated (captured), conditioned, compressed, and transported to a storage location for long-term isolation from the atmosphere. Sometimes referred to as *carbon capture and storage*. See also carbon dioxide capture and utilization (CCU) and bioenergy with carbon dioxide capture and storage (BECCS) (IPCC 2018).
21. **CARBON (DIOXIDE) CAPTURE AND UTILIZATION (CCU)** – A process in which carbon dioxide is captured and then used to produce a new product. If the carbon dioxide is stored in a product for a climate-relevant time horizon, this is referred to as carbon dioxide capture, utilization, and storage.

Only then, and only combined with carbon dioxide recently removed from the atmosphere, can CCU lead to carbon dioxide removal. CCU is sometimes referred to as carbon dioxide capture and use (IPCC 2018).

22. **CARBON DIOXIDE EQUIVALENT (CO_2e)** – The amount of carbon dioxide emission that would cause the same integrated radiative forcing or temperature change, over a given time horizon, as an emitted amount of a greenhouse gas (GHG) or a mixture of GHGs. There are a number of ways to compute such equivalent emissions and choose appropriate time horizons. Most typically, the carbon dioxide equivalent (CO_2e) emission is obtained by multiplying the emission of a GHG by its global warming potential (GWP) for a 100-year time horizon. For a mix of GHGs, it is obtained by summing the CO_2e emissions of each gas. CO_2e emission is a common scale for comparing emissions of different GHGs but does not imply equivalence of the corresponding climate change responses. There is generally no connection between CO_2e emissions and resulting CO_2e concentrations (IPCC 2018).
23. **CARBON DIOXIDE REMOVAL (CDR)** – Anthropogenic activities remove carbon dioxide from the atmosphere and durably store it in geological, terrestrial, or ocean reservoirs, or in products. It includes existing and potential anthropogenic enhancement of biological or geochemical sinks and direct air capture and storage but excludes natural carbon dioxide uptake not directly caused by human activities (IPCC 2018). Also, *carbon removal*.
24. **CARBON EMISSIONS** – The production of a greenhouse gas (GHG) or a group of GHGs, aerosols, and GHG precursors; can be due to human activities, natural processes, or the combination of the two.
25. **CARBON FARMING** – The use of specific on-farm practices designed to take carbon out of the atmosphere and store it in soils and plant material. Carbon farming practices include application of soil amendments such as compost or biochar, conservation tillage, agroforestry, whole orchard recycling, cover crops that maximize living roots, and many others (USDA Climate Hub 2021).
26. **CARBON FOOTPRINT –** A carbon footprint measures the total greenhouse gas emissions caused directly and indirectly by a person, organization, event, or product (Carbon Trust 2021).
27. **CARBON INTENSITY (CI)** – A metric of the direct and indirect effects associated with producing and using a fuel as defined by the California Air Resources Board (CARB). CI is calculated using the California Greenhouse Gases, Regulated Emissions and Energy Use in Transportation (CA-GREET) model, the Oil Production Greenhouse Gas Emissions Estimator (OPGEE), the Global Trade Analysis Project (GTAP) for indirect land use change, and the Agro-Ecological Zone Emissions Factor (AEZ-EF) for land conversion estimates, LCFS annual carbon intensity standards, or benchmarks, which reduce over time, for gasoline, diesel, and the fuels that replace them. Carbon intensity is expressed in grams of carbon dioxide equivalent per megajoule of energy provided by that fuel. CI accounts for GHG emissions associated with all steps involved in producing, transporting, and consuming a fuel—also known as a complete life cycle of that fuel (CARB 2021a).
28. **CARBON LEAKAGE** – Increases in emissions made by individuals not participating in a project. Indirect land-use change, e.g., protection of forests in one area that may lead to logging elsewhere, constitutes carbon leakage.
29. **CARBON MARKET** – A market in which a supply of carbon offset credits is sold to companies that use them to meet their voluntary or regulatory greenhouse gas emissions goals or requirements.

30. **CARBON NEGATIVE** – A term applied to processes or products when associated carbon dioxide removal from the atmosphere exceeds carbon dioxide emissions.
31. **CARBON NEUTRAL** – (See Net Zero)
32. **CARBON OFFSETS** – Reduction, avoidance, or sequestration of one metric ton of carbon dioxide or greenhouse gas equivalent.
33. **CARBON PROGRAM** – A comprehensive and recurrent service that aggregates farmers to generate and sell carbon credits from the farm.
34. **CARBON REDUCTION** – A reduction in the rate of emissions of carbon dioxide and other greenhouse gases, according to their carbon dioxide equivalence, that enter the atmosphere. *Greenhouse gas emission reduction* may also be used.
35. **CARBON REMOVAL** – The removal of carbon dioxide from the atmosphere through ecosystem processes or engineering technologies, resulting in a net decrease of greenhouse gases in the atmosphere. Also, *carbon dioxide removal*.
36. **CARBON SEQUESTRATION** – The process of storing carbon in a carbon pool (IPCC 2018). Typically refers to sequestration of atmospheric carbon dioxide. Can be through biophysical (e.g., soil carbon sequestration) or engineering (see carbon dioxide capture and sequestration) processes.
37. **CARBON SINK** – A reservoir (natural or human; in soil, ocean, and plants) where carbon dioxide is stored (IPCC 2018).
38. **CARBON STOCK** – A unit for soil carbon storage expressed as mass carbon per area and usually specified for a nominal soil depth.
39. **CARBON STORAGE** – The process of storing carbon in a pool. See also *carbon sequestration*.
40. **CLEAN ENERGY** – Energy that is produced though methods that do not release net greenhouse gases or any other pollutants. Sources of clean energy include solar, wind, geothermal, biogas, eligible biomass, and low-impact small hydroelectric sources.
41. **CLIMATE ADAPTATION** – In human systems, the process of adjustment to actual or expected climate and its effects, to moderate harm or exploit beneficial opportunities. In natural systems, the process of adjustment to actual climate and its effects; human intervention may facilitate adjustment to expected climate and its effects (IPCC 2018).
42. **CLIMATE CHANGE** – A change in the state of the climate that can be identified (e.g., by using statistical tests) by changes in the mean and/or the variability of its properties and that persists for an extended period, typically decades or longer (IPCC 2018).
43. **CLIMATE CHANGE MITIGATION** – A human intervention to reduce emissions or enhance the sinks of greenhouse gases (IPCC 2018).
44. **CLIMATE-SMART AGRICULTURE** – An approach to agricultural production emphasizing the objectives of sustainably increasing agricultural productivity and incomes, adapting and building resilience to climate change, and reducing and/or removing greenhouse gas emissions where possible (FAO 2021a).

45. **CO-BENEFITS** – The positive effects that a policy or measure aimed at one objective might have on other objectives, thereby increasing the total benefits for society or the environment. Co-benefits are often subject to uncertainty and depend on local circumstances and implementation practices, among other factors (IPCC 2018).

46. **COMET-FARM** – A USDA model that provides a carbon and greenhouse gas accounting system specific to a farmer's field; it is based on the Colorado State University models CENTURY and DayCent.

47. **COMPLIANCE MARKET** – A market for carbon offsets where entities emitting greenhouse gas above a certain threshold are required by law to cut emissions and/or purchase credits. This typically applies to airlines, industries, and power generators.

48. **CONSEQUENTIAL ANALYSIS** – An approach to estimate the indirect impacts associated with the lifecycle of a product or service.

49. **CONSERVATION AGRICULTURE** – An agricultural production system that aims to sustain production while conserving natural resources through three principles: continuous minimum mechanical soil disturbance; permanent soil organic cover with crop residues and/or cover crops; and species diversification through varied crop rotations, sequences, and associations (FAO 2021b).

50. **CREDENCE GOODS** –Goods with qualities that cannot be ascertained by consumers even after purchase (Darby and Karni 1973). For example, a carbon credit based on a claim that greenhouse gases have been sequestered from the atmosphere or emissions have been avoided through a certain process is a credence good.

51. **DECARBONIZATION** – The process by which countries, individuals, or other entities aim to achieve zero fossil carbon existence. Typically refers to a reduction of the carbon emissions associated with electricity, industry, and transport (IPCC 2018).

52. **DEDICATED ENERGY FEEDSTOCK** – Dedicated energy crops are nonfood crops that can be grown on marginal land (land not suitable for traditional crops such as corn and soybeans) specifically to provide biomass. Includes both herbaceous and wood energy crops. *Herbaceous energy crops* are nonwoody; examples include biomass sorghum, switchgrass, miscanthus, and others. *Short-rotation woody crops* are fast-growing hardwood trees, such as hybrid poplar, hybrid willow, and others, that are harvested within five to 8 years of planting (OEERE 2021).

53. **DENITRIFICATION** – The microbial mediated process that removes nitrate (NO_3^-), anaerobically reducing it to nitrite (NO_2^-), then nitric oxide (NO), then the greenhouse gas nitrous oxide (N_2O), and finally dinitrogen gas (N_2). Denitrification is inhibited by oxygen and so is more common in moist to wet environments.

54. **DIRECT AIR CAPTURE and STORAGE (DACS)** – Chemical process by which carbon dioxide is captured directly from the ambient air, with subsequent storage. Also known as *Direct Air Carbon Dioxide Capture and Storage* (DACCS) (IPCC 2018).

55. **ECOSYSTEM** – A functional unit consisting of living organisms, their non-living environment, and the interactions within and between them (IPCC 2018).

56. **ECOSYSTEM SERVICES** – Ecological processes or functions having monetary or non-monetary value to individuals or society at large. These are frequently classified as (1) supporting services such as

productivity or biodiversity maintenance, (2) provisioning services such as food or fiber, (3) regulating services such as climate regulation or carbon sequestration, and (4) cultural services such as tourism or spiritual and aesthetic appreciation (IPCC 2018).

57. **ENHANCED MINERAL WEATHERING** – Enhancing the removal of carbon dioxide from the atmosphere through dissolution of silicate and carbonate rocks by grinding these minerals to small particles and actively applying them to soils, coasts, or oceans (IPCC 2018).

58. **ENTERIC FERMENTATION** – A livestock digestive process in which microorganisms in the animal's digestive system ferment the feed consumed by the animal and produce methane as a byproduct (IPCC 2000).

59. **FUEL PATHWAY** – A renewable fuel pathway includes three critical components: (1) feedstock, (2) production process, and (3) fuel type. Each combination of the three components is a separate fuel pathway. Qualifying fuel pathways are assigned one or more codes representing the type of Renewable Identification Number (RIN) (i.e., renewable fuel, advanced biofuel, biomass-based diesel, cellulosic biofuel, or cellulosic diesel) they are eligible to generate (USEPA 2021a).

60. **GEOLOGICAL SEQUESTRATION** – The process of storing carbon dioxide in underground geologic formations. The carbon dioxide is usually pressurized until it becomes a liquid, and then it is injected into porous rock formations in geologic basins (USGS 2021).

61. **GLOBAL WARMING POTENTIAL (GWP)** – A measure of how much energy the emissions of 1 ton of a gas will absorb over a given period of time, relative to the emissions of 1 ton of carbon dioxide (CO_2). The larger the GWP, the more a given gas warms the Earth compared to CO_2 over that time period. The time period usually used for GWPs is 100 years (USEPA 2021d; also see **Appendix 11.2**).

62. **GREEN BONDS** – Fixed-income financial instruments that have positive environmental and/or climate benefits. They follow the Green Bond Principles stated by the International Capital Market Association, and the proceeds from the issuance of which are to be used for prespecified types of projects.

63. **GREENHOUSE GAS (GHG)** – Greenhouse gases are those gaseous constituents of the atmosphere, both natural and anthropogenic, that absorb and emit radiation at specific wavelengths within the spectrum of terrestrial radiation emitted by the Earth's surface, the atmosphere itself, and by clouds. This property causes the greenhouse effect. Water vapor (H_2O), carbon dioxide (CO_2), nitrous oxide (N_2O), methane (CH_4), and ozone (O_3) are the primary GHGs in the Earth's atmosphere.

64. **GREET MODEL** – GREET (Greenhouse gases, Regulated Emissions, and Energy use in Transportation) is a full life-cycle model sponsored by the Argonne National Laboratory (US Department of Energy's Office of Energy Efficiency and Renewable Energy). It fully evaluates energy and emission impacts of advanced and new transportation fuels, the fuel cycle from well to wheel, and the vehicle cycle through material recovery and vehicle disposal. The state of California uses a version of the GREET model (CA-GREET) that has been updated to calculate the carbon intensity of specific biofuel production projects as a part of the Low Carbon Fuel Standard.

65. **INDIRECT LAND USE CHANGE/INDUCED LAND USE CHANGE (iLUC)** – When land use change in one region causes land use change in another region as a result of changes in supply, iLUC can result in *carbon leakage*.

66. **IMMOBILIZATION INTO BIOMASS** – The processes where biota, at the cellular level, assimilate elements such as carbon and nutrients into their biomass. Because soil organic matter ultimately is derived from biomass, immobilization is a prerequisite for processes such as soil carbon sequestration. In the context of soils, immobilization often refers to soil microorganisms; however, plants also assimilate these elements. This process is counter to *mineralization.*
67. **INSETS** – Avoided or reduced emissions, or carbon sequestered by a company through a project within its own value chain. Emissions are upstream or downstream within the company's own value chain. Insetting can be achieved in parallel with offsetting.
68. **LIFE CYCLE ASSESSMENT (LCA)** – A standard technique for the accounting of inputs and outputs of materials and energy and the associated environmental impacts attributable to a product or service system throughout its life cycle as defined by the ISO 14001 international standard.
69. **LOW-CARBON FUEL** – A fuel source that meets a *low-carbon fuel standard*.
70. **LOW-CARBON FUEL STANDARD (LCFS)** – A ruling that mandates the reduction in the carbon intensity of energy systems in a given sector. The first LCFS was enacted by the California Air Resources Board (CARBb) in 2007 to cut GHG emissions and other smog-forming and toxic air pollutants by improving vehicle technology, reducing fuel consumption, and increasing transportation mobility options. Oregon, British Columbia, Europe, and other institutions have adopted or considered LCFS measures. This is an active area: many additional states in the United States and countries are actively considering clean fuel legislation.
71. **MACRO FUNDING** – Public financial loans or grants and funding of projects, such as the 2008 recovery package in the United States.
72. **MATERIAL FLOW ANALYSIS (MFA)** – The study of physical flows of natural resources and materials into, through, and out of a given system (usually the economy). It is based on methodically organized accounts in physical units and uses the principle of mass (or energy) balancing to analyze the relationships between material/energy flows, human activities (including economic and trade developments), and environmental changes (OECD 2008).
73. **MAXIMUM RETURN TO NITROGEN (MRTN)** – The nitrogen application rate where the economic net return of nitrogen application is maximized (ISU 2021).
74. **MEASUREMENT** – Quantification of the sources and removals of greenhouse gases. These sources and removals include emissions, emissions reductions, and/or removal of carbon dioxide from the atmosphere (UNFCCC 2014). More specifically at the farm level, measurement involves estimation of carbon credits based on farm management data combined with field-based sampling, modeling, a combination of sampling and modeling, or sampling combined with remotely sensed data. The credits are recorded by a carbon registry.
75. **MEASUREMENT, REPORTING, AND VERIFICATION (MRV)** – A system or protocol for tracking specific methods or outcomes, transparently communicating specific information, and validating that the information is accurate and complete (Oldfield et al. 2021).
76. **METHANE (CH_4)** – One of the six greenhouse gases to be mitigated under the Kyoto Protocol. Methane is the major component of natural gas and associated with all hydrocarbon fuels.

Significant emissions occur as a result of animal husbandry and agriculture, and their management represents a major mitigation option (IPCC 2018).

77. **MICRO INCENTIVES** – Programs to support adoption of practice by providing a financial incentive, such as tax benefits or penalties, insurance discount, cost-share, or payment for practice implementation.
78. **MINERALIZATION OF ORGANIC MATTER** – Processes acting on organic matter that convert elements, such as carbon, nitrogen, and phosphorus, from their organic forms to inorganic forms (e.g., organic carbon to carbon dioxide) and release them to the gas phase or to the solution phase. This process is counter to *immobilization.*
79. **MORAL HAZARD** – A situation when an individual can take advantage of a deal or situation, knowing that all the risks and fallout will have to be borne by another party (CFI 2021).
80. **NEGATIVE EMISSION TECHNOLOGIES (NETs)** – Systems that achieve net carbon removal from the atmosphere with the optional production of a marketable good or service. For example, carbon negative cement or direct air capture (DAC) technology that mineralizes atmospheric carbon using alkaline waste streams.
81. **NET ZERO** – Net zero emissions are achieved when anthropogenic emissions of greenhouse gases to the atmosphere are balanced by anthropogenic removals over a specified period. Where multiple greenhouse gases are involved, the quantification of net zero emissions depends on the climate metric chosen to compare emissions of different gases (such as global warming potential, global temperature change potential, and others, as well as the chosen time horizon) (IPCC 2018).
82. **NITROGEN UTILIZATION EFFICIENCY (NUE)** – General reference to a crop response to nitrogen fertilizer. There are many ways to calculate NUE. One common approach is to calculate the ratio of crop nitrogen uptake to the total input of nitrogen fertilizer.
83. **NITROUS OXIDE (N_2O)** – One of the six greenhouse gases to be mitigated under the Kyoto Protocol. The main anthropogenic source of nitrous oxide is agriculture (soil and animal manure management), but important contributions also come from sewage treatment, fossil fuel combustion, and chemical industrial processes. Nitrous oxide is also produced naturally from a wide variety of biological sources in soil and water, particularly microbial action in wet tropical forests.
84. **PERMANENCE** – Permanence is the principle that emissions reductions should be enduring and there should be technical or contractual protections that ensure emissions reductions or sequestration is not reversed over a meaningful period of time, thus preserving the climate benefits of an activity.
85. **POOL** – A conceptual part of an environment where specific materials reside. For example, major carbon pools include the atmosphere, the ocean, soil, biomass, minerals, etc.
86. **PRACTICE ADOPTION** – Practice adoption implies a commitment or expectation to continue using the practice in the long term. Adoption is often discussed as a binary concept: adoption versus non-adoption; however, farm-level adoption rate is a continuous variable in terms of the number of farmers who adopt a practice to any extent, from zero to 100%, based on the percentage of land on which a practice is used.

87. **REGENERATIVE AGRICULTURE** – Agricultural practices aimed at promoting soil health by restoring soil's organic carbon (note: definitions vary) (Ranganathan et al. 2020).
88. **RENEWABLE DIESEL (VERSUS BIODIESEL**) – Renewable diesel fuel, sometimes called green diesel, is a biofuel that is chemically the same as petroleum diesel fuel. Renewable diesel meets the American Society for Testing and Materials (ASTM) specification ASTM D975 for petroleum diesel and may be used in existing petroleum pipelines, storage tanks, and diesel engines. It can be produced from cellulosic biomass materials, such as crop residues, wood and sawdust, and switchgrass, and it qualifies as an advanced biofuel under the Renewable Fuel Standard (RFS) Program (USEIA 2021).
89. **RENEWABLE ENERGY** – Energy that is generated from natural processes that are naturally replenishing but flow-limited; renewable resources are virtually inexhaustible in duration but limited in the amount of energy that is available per unit of time. This includes sunlight, geothermal heat, wind, tides, water, and various forms of biomass.
90. **RENEWABLE ENERGY CREDIT (REC)** – A renewable energy certificate (REC) is a tradeable, market-based instrument that represents the legal property rights to the "renewable-ness"—or all non-power attributes—of renewable electricity generation. A REC can be sold separately from the actual electricity (kilowatt-hour, or kWh). The REC owner has exclusive rights to make claims about "using" or "being powered with" the renewable electricity associated with that REC. A REC is issued for every megawatt-hour (MWh) of electricity generated and delivered to the electric grid from a renewable energy resource (USEPA 2021c).
91. **RENEWABLE IDENTIFICATION NUMBER (RIN)** – Renewable identification numbers (RINs) are credits used for compliance and are the "currency" of the United States Environmental Protection Agency's Renewable Fuels Standard (RFS) program. Renewable fuel producers generate RINs. Market participants trade RINs. Obligated parties obtain and then ultimately retire RINs for compliance (USEPA 2021e).
92. **RENEWABLE NATURAL GAS (RNG)** – Renewable natural gas (RNG) is a term used to describe biogas that has been upgraded to replace fossil natural gas. The biogas used to produce RNG comes from a variety of sources, including municipal solid waste landfills, digesters at water resource recovery facilities (wastewater treatment plants), livestock farms, food production facilities, and organic waste management operations (USEPA 2021b).
93. **REPORTING** – Conveyance of information on performance under carbon programs.
94. **RESILIENCE** – The capacity of social, economic, and environmental systems to cope with a hazardous event or trend or disturbance, responding or reorganizing in ways that maintain their essential function, identity, and structure while also maintaining the capacity for adaptation, learning, and transformation (IPCC 2018).
95. **REVERSAL** – Loss in carbon that was previously sequestered.
96. **SCOPE 1 EMISSIONS** – Direct emissions from sources owned and/or controlled by the company; e.g., emissions from the company's facilities.
97. **SCOPE 2 EMISSIONS** – Emissions from generation of purchased energy, e.g., emissions from a company's utility provider.

98. **SCOPE 3 EMISSIONS** – Emissions from sources not owned or directly controlled by the company, e.g., emissions from a company's upstream supply chain.

99. **SECTOR SPECIFIC FOCUS** – Focus on a specific sector, such as agriculture, energy, construction.

100. **SOCIAL COST OF CARBON** – The net present value of aggregate climate damages (with overall harmful damages expressed as a number with positive sign) from one more metric ton of carbon in the form of carbon dioxide, conditional on global emissions trajectory over time.

101. **SOIL CARBON** – Consists of carbon in soil organic matter and inorganic carbon as carbonate minerals. Soil carbon can be either a carbon sink or a source in regard to the global carbon cycle, playing a role in biogeochemistry, climate change mitigation, and constructing global climate models.

102. **SOIL CARBON SATURATION** – The soil carbon saturation concept proposes that there is a maximum concentration of carbon that can be attained by a given soil. Hence, under the carbon saturation model, when a soil is near its saturation point for organic carbon, soil organic carbon concentration may increase if carbon inputs continue, but the rate of conversion to stable organic carbon will decrease. Hence the storage efficiency decreases toward zero as the soil carbon content approaches the saturation level, at which point newly added carbon will be lost to mobile gas or aqueous phases in the soil (Stewart et al. 2007). This concept may or may not be relevant for Iowa soils.

103. **SOIL HEALTH** – The continued capacity of soil to function as a vital living ecosystem that sustains plants, animals, and humans. Also referred to as *soil quality* (USDA NRCS 2021).

104. **SOIL INORGANIC CARBON** – The fraction of total soil carbon that is found in carbonate minerals, both as solids such as calcium carbonate ($CaCO_3$), and in soluble form, such as bicarbonate (HCO_3^-).

105. **SOIL ORGANIC CARBON (SOC)** – The fraction of total soil carbon that is found in soil organic matter. Soil organic carbon is found in highly heterogeneous organic compounds owing to different sources (e.g., plant litter, root exudates), stages of and susceptibility to decomposition by biota, and interactions with abiotic particles (e.g., bonding to soil mineral surfaces).

106. **SOIL ORGANIC MATTER (SOM)** – Portion of soil containing decomposed plant, animal, and microbial material. Soil organic matter is 50–60 % carbon (typically approximated at 58% carbon) but also contains the elements hydrogen, oxygen, nitrogen, sulfur, and phosphorus.

107. **TECHNO ECONOMIC ANALYSIS (TEA)** – A methodology that evaluates whether an existing or proposed chemical process is commercially competitive based on engineering and economic considerations. The United States Department of Energy employs TEA to evaluate portfolios of energy technology. The National Energy Technology Laboratory offers guidelines for the application of TEA to energy and fuel system.

108. **THIRD-PARTY VERIFICATION** – An entity that is independent of the provider or purchaser of carbon/greenhouse gas credits such that they can provide documentation that the practice associated with the credits is being implemented per the agreement. These entities hold a certification, accepted by the market, that they are a credible source for this verification.

109. **TRADEOFFS** – The negative effects that a policy, measure, or action aimed at one objective might have on other objectives. As the contrary outcome to a co-benefit, a tradeoff may incur either

total net positive or negative benefits for society or the environment dependent upon the values of the objectives involved. Tradeoffs are often subject to uncertainty and depend on local circumstances and implementation practices, among other factors.

110. **UNCERTAINTY** – A quantitative measurement of variability in the data. In other words, uncertainty in science refers to the idea that all data have a range of expected values as opposed to a precise point value. This uncertainty can be categorized in two ways: accuracy and precision.

111. **VERIFICATION** – The process by which an accredited third-party verifier examines or reviews a project, including the methodology and emission reduction or removal calculations, that the regenerative practices are occurring on the farm and that soil organic carbon is being properly accounted for (Oldfield et al. 2021).

112. **VOLUNTARY MARKET** – A market for carbon offsets where entities opt to cut emissions and/or purchase credits but are not required to do so by regulation. Examples include corporations and governments that purchase credits based upon a policy goal.

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Appendix 11.2 Primer on Calculating and Communicating Global Warming Potential

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Greenhouse gases (GHGs) are gaseous constituents of the atmosphere that absorb and emit radiation at specific wavelengths within the spectrum of terrestrial radiation emitted by the Earth's surface, the atmosphere itself, and by clouds (IPCC 2018). These properties cause the greenhouse effect. GHGs can be of natural or anthropogenic origin. Water vapor (H_2O), carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), and ozone (O_3) are the primary GHGs in the Earth's atmosphere. Carbon dioxide, methane, and nitrous oxide are GHGs relevant to agriculture-related carbon initiatives.

The impacts of GHGs are often compared or summed to respectively compare or understand their global warming potential (GWP). GWP is a measure of how much energy the emissions of 1 metric ton of a gas will absorb over a given period of time after a short-term pulse of emission, relative to the emissions of 1 metric ton of carbon dioxide. The larger the GWP, the more a given gas warms the Earth compared to carbon dioxide over that time period. The measurement unit associated with GWP is carbon dioxide equivalency, or CO_2e.

The time period often used to determine GWP is 100 years, expressed as GWP-100. There are numerous ways, however, to compute GWP and assess the impacts of GHGs over time (e.g., GWP-20, GWO-500, GWP*, GWP with or without climate-carbon feedbacks). The calculation used depends on an individual's or group's goal(s).

According to the United States Environmental Protection Agency (USEPA 2021):

- Carbon dioxide, by definition, has a GWP of 1 and lasts in the atmosphere for thousands of years.
- Methane has a GWP of 28–34 over 100 years. The GWP accounts for both the greater potency of methane as a heat-trapping gas as well as its shorter atmospheric lifetime of approximately 12 years compared to carbon dioxide.
- Nitrous oxide has a GWP of 265–298 over 100 years. The GWP accounts for both the greater potency of nitrous oxide as well as its shorter atmospheric lifetime of approximately 114 years compared to carbon dioxide.

These values represent the range of the lowest to the highest values listed in the Intergovernmental Panel on Climate Change (IPCC) Fifth Assessment Report (AR5), published in 2013 (IPCC 2013). GWP ranges for methane and nitrous oxide reflect changes in scientific knowledge as well as different methods for calculating GWP over a 100-year time horizon. The IPCC presents multiple methods for calculating GWPs based on whether climate-carbon feedbacks (or the influence of future warming on the carbon cycle) are accounted for.

The IPCC provides updated assessments that reflect changes in scientific knowledge. The IPCC's calculations of GWP have changed over time based on improved scientific measurement, modeling, and understanding. The GWP-100 value for methane was 25 in the fourth assessment (AR4; IPCC 2007) and 28–34 in the fifth assessment (AR5; IPCC 2013). Similarly, the GWP-100 of nitrous oxide was 298 in the fourth assessment and 265–298 in the fifth assessment. Additional updates to GWP are expected in the sixth assessment as well as for methane from fossil fuels sources to be calculated separately from methane from biogenic sources (IPCC 2021). The sixth assessment has been released but not been officially published; minor modifications may yet be made.

The EPA's *Inventory of US Greenhouse Gas Emissions and Sinks* complies with international GHG reporting standards under the United Nations Framework Convention on Climate Change (UNFCCC). UNFCCC guidelines now require the use of GWP values from the IPCC's Fourth Assessment Report (AR4), published in 2007. EPA's inventory also presents information so that carbon dioxide equivalents (CO_2e) can be calculated using any GWPs. Data collected by EPA's Greenhouse Gas Reporting Program are used in the Inventory, so the Reporting Program generally uses GWP values from the AR4. The Reporting Program collects data about some industrial gases that do not have GWPs listed in the AR4; for these gases, the Reporting Program uses GWP values from other sources, such as the IPCC's Fifth Assessment Report. EPA's voluntary methane reduction programs also use GWPs from the AR4 report for calculating methane emissions reductions through energy recovery projects to maintain consistency with the national emissions presented in the Inventory.

Carbon registries are following the guidance provided by the IPCC. For example, the Gold Standard currently uses GWP values from the IPCC's Fifth Assessment Report, but states that the technical advisory committee will reevaluate in the future based on the IPCC's guidance (Gold Standard 2021).

We use multiple GWP values in this report, depending on the publication being cited or what the respective authors used in their work. Note, however, that the different GWPs for the same gas are typically within the range of uncertainty on their values.

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Appendix 11.3 Crop Yield Prediction

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Because in-field measurement is labor and time intensive and costly, carbon programs that buy/sell agricultural carbon credits depend on agroecosystem models to predict changes in GHG (greenhouse gas) emissions and soil organic carbon (SOC) stocks and with changes in farming practices. Agroecosystem models are able to do some things well at present; for example, predicting potential maximum yields. Other measures, such as yield variability, are challenging, especially at the field or practice scale needed to account for carbon credits.

The table below summarizes the current state of scientists' ability to predict crop yield, including goals, current state, and future needs of process-based models used for yield prediction. This information is illustrative because yield is easier to measure in the field and has received the most field measurement compared to all other variables incorporated in agroecosystem models. Other agroecosystem outcomes, including GHG emissions and soil carbon storage, are mediated by similar environmental, genetic, and management factors, but have received far less scientific attention. Predictions of the variables of interest to carbon markets—GHG emissions and soil carbon storage—are less reliable because field measurements to support them are sparser.

Goals for predicting within-field yield	Current state	Future needs and/or challenges
Predict this year's within-field yield for a field with in-hand yield monitor data	Yield can be predicted mid-season using a combination of remote sensing data such as satellite imagery or elevation; however, these predictions are still highly variable in accuracy depending on the field, capturing anywhere from 10–80% of variation in yield (Skakun et al. 2021). It is generally harder to predict yield in high-performing fields with little variation. Predictions are best at identifying lowest yield parts of fields and fields with more variation in yield. Yield monitor data or a combination of remote sensing and prior years' calibrated yield monitor data (Maestrini and Basso 2018a) are best at identifying and explaining causes of yield variation.	Need better remote sensing data such as high-resolution imagery, elevation, etc. Need high quality, calibrated yield monitor data for various management practices, topographies, and climates for multiple years (4+). It does not appear that a majority of farmers are currently calibrating yield monitor data, and farmers on poorer ground are more likely to not have a yield monitor at all (Nowatzke et al. forthcoming).

Predict future years' within-field yield for a field with in-hand yield monitor data	Future yields can be partially predicted with prior knowledge, such as field location, and topographic characteristics, such as soil type and elevation (Maestrini and Basso 2018b). The other source of making predictions can be prior years' yield monitor data, though the number of years needed to detect trends can vary greatly (4 to 20 years) (Bunselmeyer and Lauer 2015). This also requires yield monitor equipment to be calibrated.	Many other unknown factors can change final harvested yield. Need high quality, calibrated yield monitor data for various management practices, topographies, and climates for multiple years (4+).
Predict this year's yield for a field without in-hand yield monitor data	Using one field's data to make predictions about another field depends on how similar they are in location, topography, yield distribution, and management (Schwalbert et al. 2018). This is, in general, a more challenging task. It is possible that enough years of high quality, calibrated yield monitor and management data for a spatially distributed representation of the target area would allow for accurate in-field predictions; however, this would still rely on remote sensing data as well, and still encounter those same limitations.	Need high quality, calibrated yield monitor data for various management practices, topographies, and climates for multiple years (4+), spatially represented throughout Iowa. Still hindered by remote sensing variation in accuracy.
Predict future years' yield for a field without in-hand yield monitor data	This task is dependent on improving the ability to predict this year's yield for an unknown field (see previous row); if successful, multiple years of prior yield monitor data for surrounding fields could be taken, along with multiple years of remote sensing data for the target field to make future years' yield predictions for the new field. This appears to be relatively untested for Iowa though.	Contains all the challenges of prior goals, with more data coordination. Relatively untested for Iowa.

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Appendix 11.4 Methods for Determining Methane and Nitrous Oxide Emissions Associated with Animal Agriculture

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A variety of tools are used to directly measure emissions of methane from enteric and manure sources, but the majority of these tools are research-based. Models and estimates based on standard reference sources are also widely used. An example complication that must be considered is the difference in feed and emissions from lactating versus non-lactating dairy cattle. Standard references and models may not represent Iowa conditions generally, or important farm-to-farm differences in Iowa, and more Iowa data are needed.

Determining Enteric Methane Emissions

Measurement of methane emissions from animals requires complex and expensive equipment or techniques, such as respiration chambers and the sulfur hexafluoride (SF6) tracer technique. Therefore, prediction models are used widely to estimate methane emissions from animals for national or global GHG inventories (Kebreab et al. 2008). Several mathematical models have been developed to predict enteric methane emissions from livestock, including both empirical and mechanistic models. The empirical models have provided a better alternative in practical situations because they are simpler and easily applicable in terms of information and computer software requirements (Appuhamy et al. 2016). Nonetheless, most of the available empirical models seem to suffer from narrow spatial focus and limited observations. Those models might not be able to predict the emissions in every region with equal accuracy (Hristov et al. 2018). Appuhamy et al. (2016) evaluated the accuracy of 40 empirical models, including the IPCC (2006) Tier II model, for predicting enteric methane emissions from dairy cows (g per cow per d) using data specific to North America, Europe, and Oceania. They found a different set of models predicting the emissions accurately in each region, which highlights the need for using region-specific models. In their evaluation, five other models more accurately predicted methane emissions of the United States and Canada than the IPCC (2006) Tier II model. The 2019 refinements to the IPCC (2006) models have addressed the regional differences to some extent by accounting particularly for the variability in diet composition (IPCC 2019). Nonetheless, the application of IPCC (2019) models still involves a cascade of calculations requiring an extensive amount of data that are hard to find on commercial farms. Niu et al. (2018) developed a set of models to predict enteric methane emissions from dairy cows using a large intercontinental dataset. They concluded that simple models that consider only DMI of individual cows (kg per cow per d) provide a good prediction of enteric methane emissions (g per cow per d) in the United States. This is not surprising as variability in DMI can explain 60–70% of the variability in enteric methane production of cows (Appuhamy et al. 2016). Nonetheless, measuring the DMI of individual animals is rather challenging in commercial farms. Alternatively, the DMI can be calculated using readily available information such as milk yield and the stage of lactation of individual cows in commercial farms. Appuhamy et al. (2016) showed that the calculated DMI could predict enteric methane emissions of cows in the United States and Canada as accurate as the measured DMI. Moreover, the literature offers several empirical models to predict methane emissions from non-lactating cows and dairy heifers (Moraes et al. 2014). Again, those models also require DMI to predict the emissions. The good news is the literature (Hayirli et al. 2003; Hoffman et al. 2008) also provides models to project DMI of those dairy cattle groups.

In line with Appuhamy et al. (2016), van Lingen et al. (2019) emphasized the importance of using region-specific models for predicting enteric methane emissions from beef cattle. They evaluated several empirical models and concluded that a model including DMI and dietary ether extract content best predicted the enteric methane emissions from North American cattle consuming high-grain diets (e.g., the feedlot cattle). The model's prediction error was about 50% of the error of the IPCC Tier 2 model (IPCC 2006). However, the IPCC model and a model based on DMI alone (Charmley et al. 2016) best predicted enteric methane emissions of beef cattle on pasture or high-forage diets (van Lingen et al. 2019). Again, application of those models also requires DMI of individual animals, which can be calculated using the equations in NRC (1996) or Anele et al. (2014) or the methodology in IPCC (2019). Relatively few models are available to predict enteric methane emissions of swine. The IPCC uses a constant emission factor of 3.3 lb per animal per year (1.5 kg per animal per year) for any group of swine (growing, finishing, or breeding) even though the enteric methane production can be highly variable depending on DMI and diet. Jørgenson et al. (2011) developed equations to predict enteric methane emissions from swine depending on fermentable fiber intake and were able to account for the variability among different swine groups. Nonetheless, those models have not been evaluated properly for prediction accuracy.

In summary, while there are several empirical models to predict enteric methane emissions from livestock in Iowa, the ability to determine DMI is critical for achieving accurate and efficient predictions from those models. Therefore, methane production and DMI measurements representative of Iowa livestock are needed to evaluate those models for their predictive power. Powerful models can be used to determine baseline enteric methane emissions relative to which the mitigation practices can be evaluated.

Determining Methane and Nitrous Oxide Emissions from Manure

There are currently two methods for estimating methane production from livestock manure. The first method, the Intergovernmental Panel on Climate Change (IPCC) Tier 2 approach, requires specific waste characteristics, including the maximum methane-production capacity of the manure (B_0), the mass of volatile solids excreted from the animals, and a methane conversion factor (IPCC 1996; IPCC 2000). The B_0 typically needs to be determined using the biochemical methane production (BMP) potential assay as described by Moody et al. (2011), although the default values (e.g., 0.48 m^3 per kg of VS for swine manure) are often used in the Tier 2 approach. The ASABE standard ASAE D384.2—Manure Production and Characteristics (ASAE 2005) is used to estimate the mass of volatile solids excreted for various livestock species. It offers a set of equations to predict volatile solids output (kg per animal per day) of dairy cattle, beef cattle, swine, and poultry. Those equations can provide representative estimate of VS as they account for the variability of volatile solids driven by feed intake, dietary nutrient composition, and animal performance. The ASAE also provides default values of volatile solids output for each livestock category if required information is not available to apply the equations.

Continuing with the swine production example, the value currently listed for finishing swine is 0.56 lb (0.27 kg) of volatile solids per day per animal. Additionally, the methane conversion factor (MCF) needs to be estimated based on IPCC suggestions for the specific manure storage type and the environmental temperature (IPCC 2019). For instance, according to the updated IPCC guidelines (IPCC 2019), 31%, 55%, and 80% of MCF are applied for long-term (12 month) pit manure storage at cool, moderate, and warm temperatures, respectively. Although this methodology seems to offer a good approximation for an average animal facility, researchers have pointed out substantial variations in methane emissions (Haeussermann et al. 2006). Any practice designed to reduce methane emissions will be most beneficial when targeting operations with the most significant methane emissions.

IPCC Tier 2 approach for estimating nitrous oxide emissions from manure is similar to the approach for methane emissions (IPCC 2006). It requires two pieces of information: 1) Nitrogen excreted in manure (feces and urine combined) by individual animals, and 2) the nitrous oxide emission factor. Nitrous oxide emission factors are estimates of kilogram of nitrous oxide produced per kilogram of nitrogen excreted. These factors vary by waste management system. The IPCC offers a set of default values for manure nitrogen excretions by different animal species but encourages the use of country-specific estimates if possible (IPCC 2019). The ASAE (2005) provides US-specific default values of manure nitrogen excretions and equations to predict manure nitrogen excretions of different animal categories allowing more representative estimate for a given region or state within the United States. Others (Appuhamy et al. 2018; Reed et al. 2015; Vu et al. 2009) also provide equations to predict volatile solids and nitrogen excretions of cattle and swine.

Another method commonly used to estimate emissions from livestock facilities uses an emission factor approach. Livestock operations used to establish emission factors for the IPCC method are expected to be representative of a given animal industry (e.g., Iowa swine industry). Emission factors can also be determined at high levels of accuracy using barn-scale methodologies (Wang et al. 2010). However, these procedures are labor intensive, require substantial investment in time and equipment, and thus are not practical for assessing large numbers of farms (Jungbluth et al. 2001). In addition, researchers measuring methane emission in the field have found significant variations in emission amounts at the animal housing level (Haeussermann et al. 2006), caused in part by broad diurnal and seasonal variation (Hartung 1998). Therefore, methane or nitrous oxide emission factors will invariably have significant uncertainties when applied to specific farms. Consequently, many livestock operations in question need to be surveyed for greater confidence in the emission factors used, which may not be practical given the cost of performing these studies. Overall, cheaper alternative procedures that are capable of high throughput analysis are needed.

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