APPENDIX A:
Greenhouse Gas Analysis Methods & Adoption Scenario Development

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PART I. GHG ANALYSIS DETAILS AND METHODS

1. *INTENT AND GOALS*

This analysis is intended to provide a high-level quantification of the climate change mitigation potential of "conservation practice" implementation in annual systems, the conversion of annual cropland to perennial crops and covers in Wisconsin, drawn from currently available CO2 offset values published in the scientific literature, and greenhouse gas emission reductions from improved fertilizer and manure management. Through this analysis we aim to provide an accessible summary of the climate change mitigation potential for these practices as reported in the most up-to-date scientific literature, highlighting the relative efficacy of different practices, and illustrating what it will take to reach net-zero emissions in the agricultural sector.

This fills a need to explore agricultural NCS practices at a state-level. Published estimates for climate change mitigation potential on agricultural land are currently only available at the global or national level (e.g., Griscom et al. 2017, Fargione et al. 2018, Walton Family Foundation 2022). Nature4Climate's United State NCS Mapper applies the sequestration and emissions factors from a global analysis (Griscom et al. 2017) to individual states to provide a state-level estimate. While this is helpful, a single global/national value may not accurately reflect the specific circumstances in Wisconsin, since the climate change mitigation potential of practices is highly site- and context-specific. Indeed, this important limitation is acknowledged by the Nature4Climate mapper, which encourages "more detailed analysis at the state level for policy and planning purposes."

Similarly, the Carbon Reduction Potential Evaluation (CaRPE) tool provides interactive quantification of some agricultural practices at a state (and county) level. However, this tool is utilizing only a single estimate (the COMET-Farm estimate) of the climate change mitigation potential of the modeled practices. While this model does provide useful information, it has its own important limitations in that it has significant field validation gaps and only models the top 30 cm of the soil, which likely overestimates the soil carbon sequestration potential of several practices.

Given the highly variable nature of the climate change mitigation potential of these practices, it is valuable to have an analysis that can provide a range of mitigation potentials that users can tailor and interpret in the context of their own specific circumstances. This flexibility also allows our modeling approach to be easily adaptable to other states in the region that have different circumstances.

The main goals are to provide a science-based foundation for discussions and decisions about how such practices could or should be encouraged or incentivized in Wisconsin in the context of climate change mitigation by:

1) Clearly demonstrating the challenge of achieving net-zero agriculture by presenting a quantitative analysis;

2) Demonstrating the relative efficacy of different agricultural practices at sequestering carbon or reducing GHG emissions in the agricultural sector;

3) Providing an evidence-based and transparent quantification that anyone can interpret, modify, replicate, and update as new information becomes available. Many existing climate mitigation analyses are inflexible, black box analyses that are difficult to interpret and modify to better reflect a more specific geography.

2. *GENERAL APPROACH*

The baseline agricultural sector emission inventory for this analysis comes from the Wisconsin Department of Natural Resources' 2021 Greenhouse Gas Inventory (Wisconsin Department of Natural Resources 2021), which in turn uses the EPA's state inventory tool (SIT). In this inventory, the agricultural sector module includes emissions largely from the following: enteric methane emissions; manure storage methane and N_2O emissions, and N_2O emissions from fertilizer and manure field applications and plant residues (Figure 1). Carbon emissions from liming fields and urea fertilization, as well as methane and N_2O emissions from agricultural burning are also included with minimal contributions (less than 2% of total sector emissions, combined in Wisconsin).

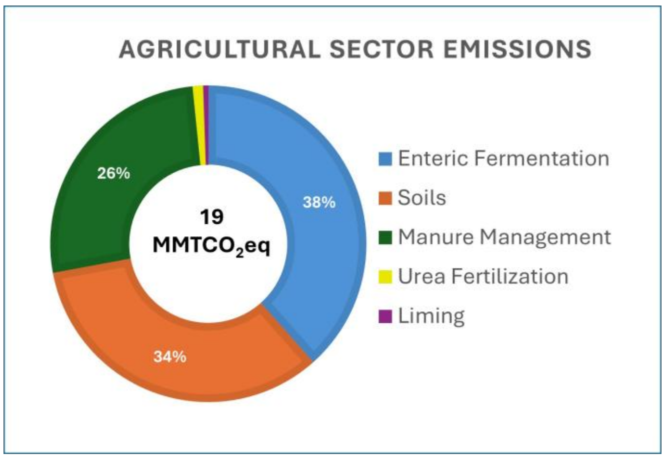

Figure 1. Breakdown of agricultural sector emissions in the Wisconsin Department of Natural Resources' greenhouse gas inventory.

The agricultural sector emissions do not include on-farm fuel and electricity use or carbon flux from the soil. These are included in various other modules of the SIT, which are ultimately synthesized together for the total state inventory.[1] Carbon flux from agricultural soils is considered in a separate Land Use, Land Use Change and Forestry (LULUCF) module. These emissions are calculated in the SIT using a lookup table of values produced using DAYCENT modeling. The agricultural soil carbon flux is reported as the combined flux of land converted to grassland,

[1] The SIT contains 11 modules: agriculture, CO_2 from fossil fuel combustion, coal, electricity combustion, industrial processes; land use, land-use change and forestry; mobile combustion; natural gas and oil; solid waste; stationary combustion; and wastewater.

grassland remaining grassland, cropland remaining cropland, and land converted to cropland. Modifying these values is beyond the scope of this analysis.

However, our goal with this analysis is to examine to what extent climate-smart agricultural practices can reduce or offset the 19 MMT from agricultural sector emissions as defined in this existing inventorying approach, and thus the current soil carbon flux is not relevant. For field management practices like cover crops, no till and conversion from annual row crops to perennial systems that could potentially sequester carbon in soil or biomass, we use existing model estimates and searched published literature for studies that reported sequestration benefits of the practice relative to an annual crop reference system. We then credit the carbon sequestration for newly adopted practices against the current agricultural sector emissions. Practices that continue to lose soil carbon have the same effect in our analysis as a practice that holds soil carbon steady since both practices have zero potential to offset agricultural sector emissions. However, we note that this overlooks the climate mitigation potential of practices that slow or the release of carbon relative to the current annual row cropping system (e.g., Dietz et al. 2024), even if it does not sequester carbon that can offset some agricultural sector emissions.

In our quantification of the potential of agricultural practices to mitigate climate change, we follow the approach used in prior evaluations at global (Griscom et al. 2017) and national scales (Fargione et al. 2018; Drever et al. 2021; Walton Family Foundation Report). Generally, mitigation potential is calculated as:

$$\textit{Mitigation Potential (tons } CO_2eq\ yr^{-1}\textit{)} = \textit{Mitigation Flux x Potential Extent of Practice Adoption}$$

Mitigation flux refers to the rate of climate mitigation per unit (e.g., soil carbon sequestration per hectare or methane reduction per ton of manure produced). Potential extent of practice adoption refers to the total adoption potential (e.g., total acres of cover crop adoption or percent of total manure produced).

We rely on published or previously-used estimates most appropriate to Wisconsin to identify the mitigation flux we use in our calculations, as detailed in the following sections. Climate mitigation fluxes can be highly variable and context-dependent. Thus, to increase confidence in flux values used in our quantification, to the greatest extent possible we rely on values reported in meta-analyses and literature reviews that pool results from multiple studies to report overarching trends across individual studies, thus minimizing the effect of a single study's limitations or bias. Where the meta-analyses provide subsets of results (e.g., specific to certain geography, climate zone or soil type) we use the subset most relevant to Wisconsin. We also supplement these larger meta-analyses with individual studies conducted in Wisconsin (or the Upper Midwest) where available.

To account for the uncertainty in the potential of various practices to mitigate climate change, we provide a range of estimates and a "best estimate" range specific to Wisconsin as detailed in the following sections.

Within the framework of this analysis, practices that do not sequester any carbon and practices that continue to lose carbon are functionally the same since they provide no net sequestration against which to reduce agricultural sector emissions. If an agricultural practice is reported to be a net soil carbon source rather than sink, we assign to it a mitigation flux of 0, rather than assigning a negative flux since the carbon flux from agricultural soils is considered in a separate inventory module, as described above.

3. *SCOPE*

The following agricultural practices are included in our quantification:

3.1 Cover cropping: cover cropping is the practice of planting crops in between primary harvested crops to keep the ground covered. This practice helps reduce erosion, improve soil health, and suppress weeds. Cover cropping can increase soil carbon by increasing carbon inputs via additional root biomass, microbial carbon transfer, and incorporation of cover crop residue upon termination.

3.2 No-till: tillage is the process of turning up the soil and incorporating any surface residue into the soil to provide a clean surface for planting. No-till refers to the practice of not using plows, discs, cultivators, etc., to invert, turn, or mix the soil; usually involves no-till drill crop establishment implements that minimal-disturbance discs that cut through surface residue to cut a narrow slot in the soil that seeds are dropped into, with press wheels that follow to close the slot. This greatly reduces soil disturbance, reducing erosion, building soil health and improving soil moisture availability. No-till can also increase soil carbon by maintaining soil stability and reducing carbon losses from microbial activity. Finally, no-till reduces the number of tractor passes on a field, reducing fossil fuel usage on the farm.

3.3 Agroforestry practices: agroforestry broadly refers to the deliberate integration of trees and woody shrubs into the agricultural landscape. Agroforestry helps to sequester carbon by increasing soil carbon through the extensive and perennial root systems and soil stability provided by the trees, increased carbon inputs into the soil through leaf litter, and through below- and aboveground carbon sequestration in the woody biomass of the trees.

We are including the following agroforestry practices relevant to Wisconsin in this analysis:

- *Alley cropping* refers to a system of crops planted between rows of trees (Fig. 2). An example relevant to Wisconsin is planting winter wheat between rows of chestnut trees.

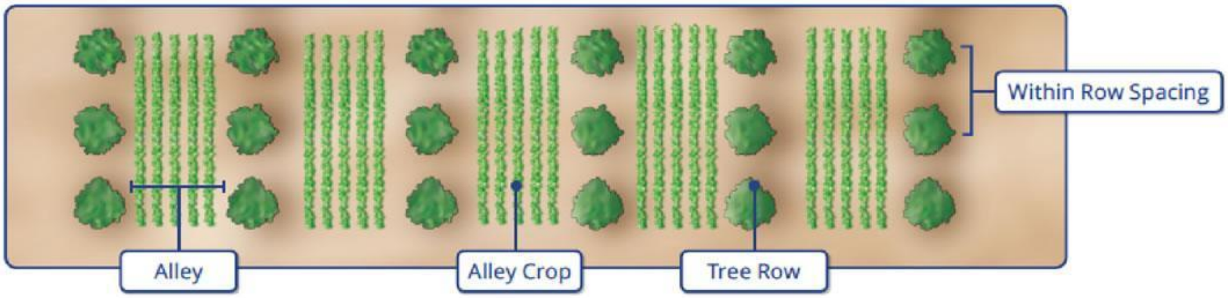

Fig. 2. Alley crop diagram. Source: USDA National Agroforestry Center Illustration https://www.fs.usda.gov/nac/practices/alley-cropping.php

- *Silvopasture* refers to the integration of trees and livestock grazing. This is accomplished by either introducing herbaceous forage into the understories of selectively thinned secondary forest fragments on existing farmland "silvopasture by exclusion"; although removal of trees will introduce additional carbon loss) or planting shade trees and windbreaks on existing, exposed pastures (silvopasture by inclusion). Only the silvopasture by inclusion was included in our analysis.

- *Windbreaks* are linear tree plantings designed to protect cropland, livestock areas, and buildings from damaging winds and snow drifts (Fig. 3).

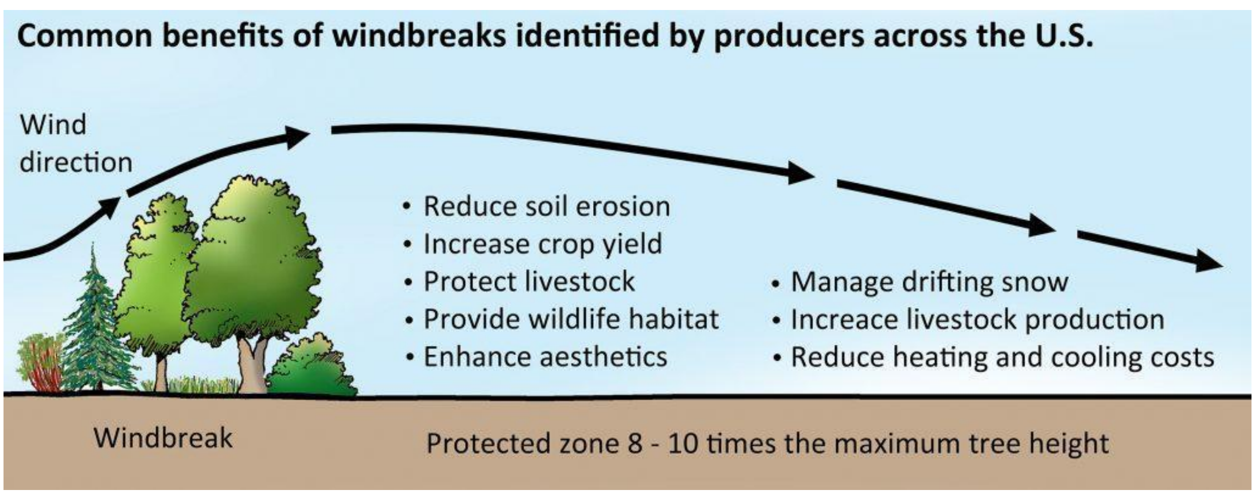

Fig. 3. Windbreak diagram Source: UW Extension: https://woodlandinfo.org/windbreaks/

- *Riparian forest buffers* are strips of trees and other woody vegetation established alongside rivers, streams and lakes (Fig. 4).

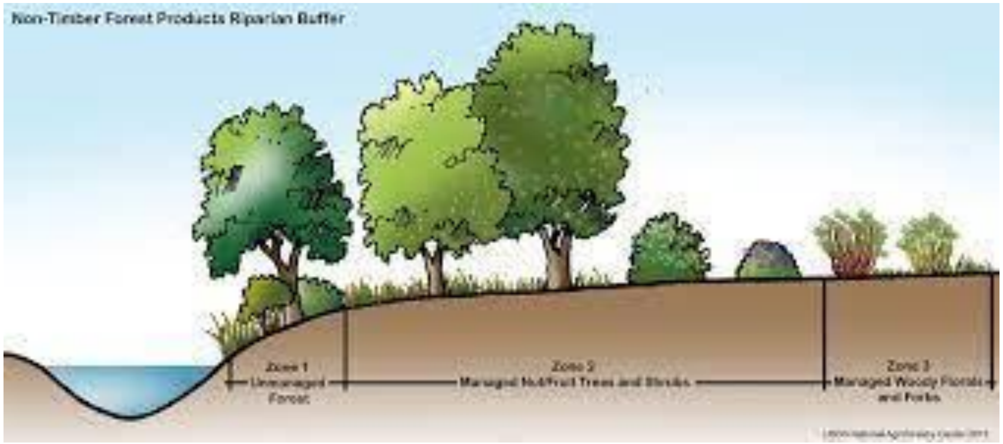

Fig. 4. Riparian buffer diagram Source: USDA forest service: https://www.fs.usda.gov/nac/practices/riparian-forest-buffers.php

3.4 Conversion from annual row crops to perennial herbaceous crops: perennial crops are crops that are planted and then maintained and harvested over multiple growing seasons, over multiple years without (or before) requiring replanting. This includes both woody crops (i.e., agroforestry), which are considered separately in this report, and herbaceous crops (i.e., grasses, legumes, oilseeds, etc.). Herbaceous perennials have multiple end-uses such as human food (e.g., Kernza®), livestock forage (e.g., alfalfa and Kernza®), and bioenergy (e.g., switchgrass). Herbaceous perennial crops provide a greater opportunity to build biomass and thus carbon inputs to the soil compared to annual crops. Perennial fields also have less soil disturbance, reducing carbon loss and promoting the stabilization of carbon in the soil.

3.5 Conversion from annual row crops to grasslands or well-managed, rotationally grazed pastures: Grasslands and well managed, rotationally grazed pastures have the potential to increase soil carbon stocks compared to annual crop fields, similar to herbaceous perennials. Rotationally grazed pastures develop robust root systems that stabilize and increase carbon inputs to the soil, stabilizing reserves of soil carbon for long-term. Grassland and well managed, rotationally grazed pastures can be managed without additional fertilization (e.g., Jackson 2022), leading to a reduction in N_2O emissions compared to annual crops grown with fertilizer input, as discussed below. Similarly, they require little or no diesel fuel to run farm equipment, so reductions in fossil fuel combustion are key climate benefits of these grasslands compared to annual row crops.

Here, we consider two transitions. First, we include the transition of annual row crops to solar farms. Large solar farms in Wisconsin are establishing deep-rooted native grasses under and around the solar panels in their vegetation management plans. As described by Walston et al. (2021), this conversion has the potential to increase soil carbon sequestration from these projects.

Second, we look at the transition from confined milk production to grassfed milk production, which will require an expansion of pastureland in the state. However, the shift from grain-fed ruminant livestock to pasture-fed ruminant livestock has numerous other effects on GHG emissions from a farm. A full discussion of this shift is discussed below in the "*Transition from Confined Dairy Production to Grazed Dairy Production*" section below.

3.6 Improved Grazing Management: Optimizing grazing intensity (i.e., not overgrazing, but not underutilizing forage production either) on existing pastures have the potential to sequester soil carbon by increasing pasture above and belowground biomass production, reducing soil erosion, and improving soil health.

3.7 Biochar soil amendments: Biological charcoal (biochar) incorporation into agricultural fields represents a potential carbon sink. When biomass like agricultural residues or wood biomass leftover from logging operations is burned or left to decompose, much of the previously-fixed carbon is released back into the atmosphere. Creating biochar from these residues through pyrolysis and then incorporating it into agricultural soils stabilizes the carbon and keeps it in the ground for hundreds—and potentially thousands—of years. In addition to sequestering carbon, incorporating biochar into agricultural fields can improve soil health and productivity.

3.8 Nitrogen management: N_2O soil emissions are produced through microbe-mediated nitrification and denitrification processes, and increased emissions are driven primarily by the addition of synthetic N fertilizers and animal manure to fields. The steady increase in atmospheric N_2O concentrations—from approximately 290 ppb in 1940 to 330 ppb in 2017—is linked to the increase in reactive nitrogen in the environment, largely due to the increased use of nitrogen fertilizers in the agricultural sector

(Thompson et al. 2019). N_2 O emissions from a field increase with increasing nitrogen inputs to the field (e.g., fertilizer applications), so improved nutrient management will reduce the amount of nitrogen inputs to fields, thus decreasing N_2 O emissions. In addition to reducing N_2O emissions from the field, reduced nitrogen fertilizer use will avoid emissions associated with its production, which is energy-intensive and a source of "upstream" greenhouse gas emissions.

Strategies to reduce emissions from agrochemical fertilizer use:

- Reduce/eliminate N fertilizer addition through conversion to perennial systems or less nitrogen-intensive crops
- Practice the 4 Rs: Right time, right place, right form, right rate
- Improved nitrogen use efficiency in crops

3.9 Manure management: Manure management is an important source of methane and N_2O emissions in Wisconsin, accounting for 25% of GHG emissions from the agricultural sector (5 MMT CO_2eq), not including the emissions from the manure when it is spread on the fields.

Methane is produced by the bacterial breakdown of volatile solids in manure under anaerobic conditions. Warm, anaerobic, water-based conditions are most conducive to methane production. N_2O is produced via a combined nitrification denitrification of the N contained in the waste. Ammonia is converted into nitrate in aerobic conditions, followed by nitrate being converted to N_2O in anaerobic conditions. Dry, aerobic systems are more conducive to N_2O emissions. The amounts of volatile solids and nitrogen in the manure depend on cow size, cow digestive physiology, and diet. In our calculations, we used the EPA and DNR quantifications which provide typical nitrogen excretion and volatile solids amounts per animal value for dairy cows in Wisconsin.

The form of manure and storage conditions are the key factors determining emissions. Generally speaking, liquid manure management promotes CH_4 emissions, while solid manure management releases proportionally more N_2O (Fig. 5). Similarly, capping or allowing crust to form on liquid storage ponds will reduce CH_4 emissions but increase

N_2O emissions. Thus, there is some tension between minimizing CH_4 emissions and N_2O emissions since shifting management to minimize one can increase the other.

The manure management practices included in the State Inventory Tool Agricultural Sector are defined by the EPA (2022) as follows:

- Pasture: The manure from pasture and range grazing animals is allowed to lie as is and is not managed.
- Daily spread*:* Manure is routinely removed from a confinement facility and is applied to cropland or pasture within 24 hours of excretion
- Solid storage: The storage of manure, typically for a period of several months, in unconfined piles or stacks. Manure is able to be stacked due to the presence of a sufficient amount of bedding material or loss of moisture by evaporation.
- Deep pit: Collection and storage of manure usually with little or no added water typically below a slatted floor in an enclosed animal confinement facility. Typical storage periods range from 5 to 12 months, after which manure is removed from the pit and transferred to a treatment system or applied to land.
- Liquid slurry: Manure is stored as excreted or with some minimal addition of water to facilitate handling and is stored in either tanks or earthen ponds, usually for periods less than one year.
- Anaerobic lagoon- Uncovered anaerobic lagoons are designed and operated to combine waste stabilization and storage. Lagoon supernatant is usually used to remove manure from the associated confinement facilities to the lagoon. Anaerobic lagoons are designed with varying lengths of storage (up to a year or greater), depending on the climate region, the [volatile solid] loading rate, and other operational factors.
- Anaerobic digestor: Animal excreta with or without straw are collected and anaerobically digested in a large containment vessel (complete mix or plug flow digester) or covered lagoon.

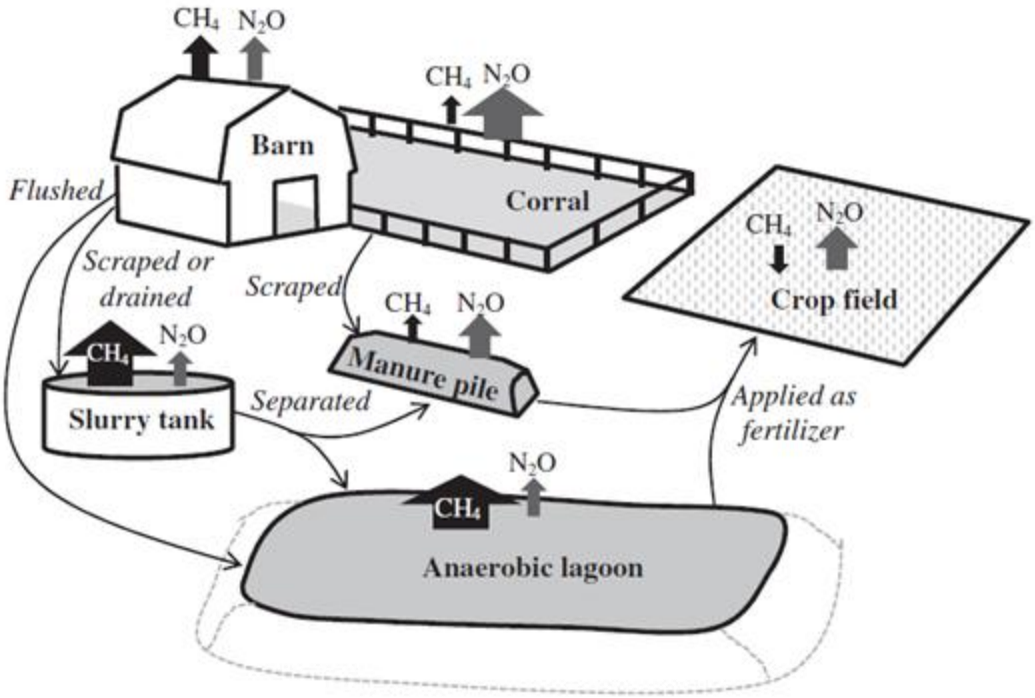

Figure 5. Relative CH_4 and N_2O emissions on dairies (source: Owen & Silver 2014). Thicker arrows represent greater emissions.

The majority of greenhouse gas emissions from manure management (i.e., not including emissions once manure is landspread) are methane emissions. GHG emissions from manure management have increased 3-fold since 1990, driven by increases in methane emissions (Fig. 6). This increase is responsible for half of the agricultural sector's emissions increase since 2005. While milk production per cow has also increased, the manure management GHG emissions per unit of milk has increased by 50% from 1990 (0.2 Mg CO_2eq per Mg milk produced) to 2018 (0.31 Mg CO_2eq per Mg milk[2]). This is largely driven by the shift away from daily spread and solid storage on smaller farms (methane conversion factor of <5%) to anaerobic lagoons and deep pits at larger farms, which create conditions that promote methane conversion (methane conversion factors of 24-68%; Figure 7).

[2] Using manure emissions from WDNR GHG inventory agricultural module and milk production data from: https://www.nass.usda.gov/Statistics_by_State/Wisconsin/Publications/Dairy/

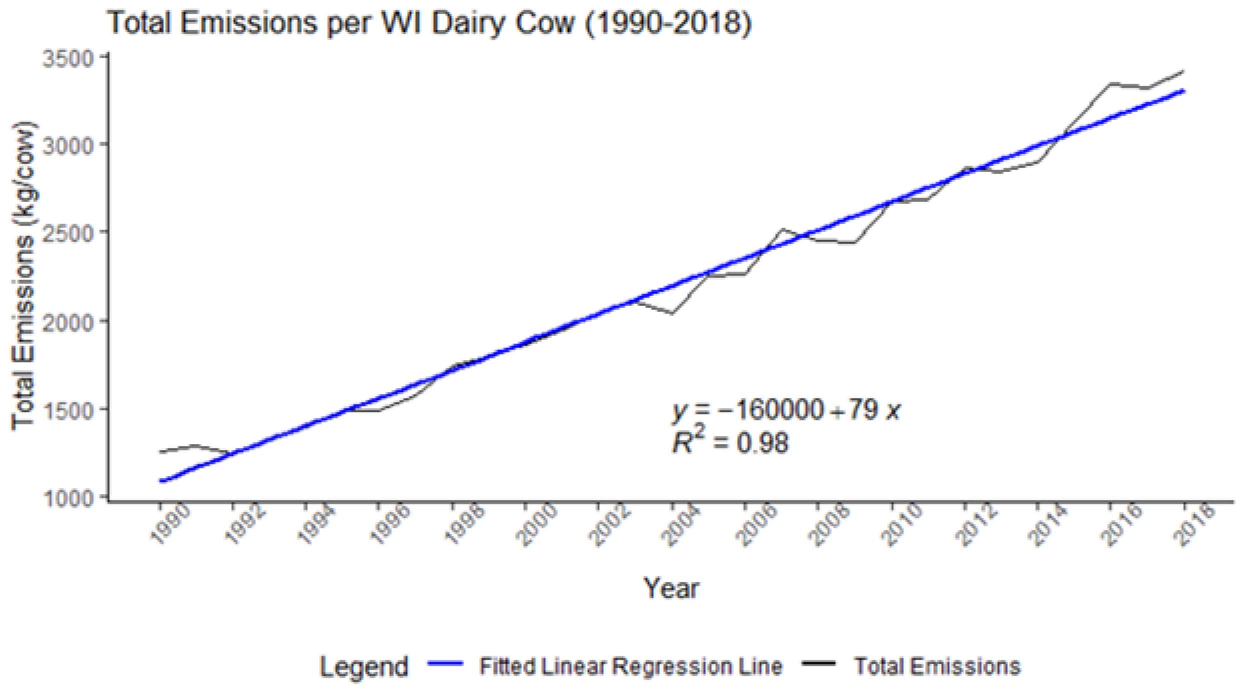

Figure 6. Manure management GHG emissions per dairy cow in Wisconsin over time, reflecting shift towards anaerobic lagoon between 1990-2018. Data from the Wisconsin DNR GHG Inventory.

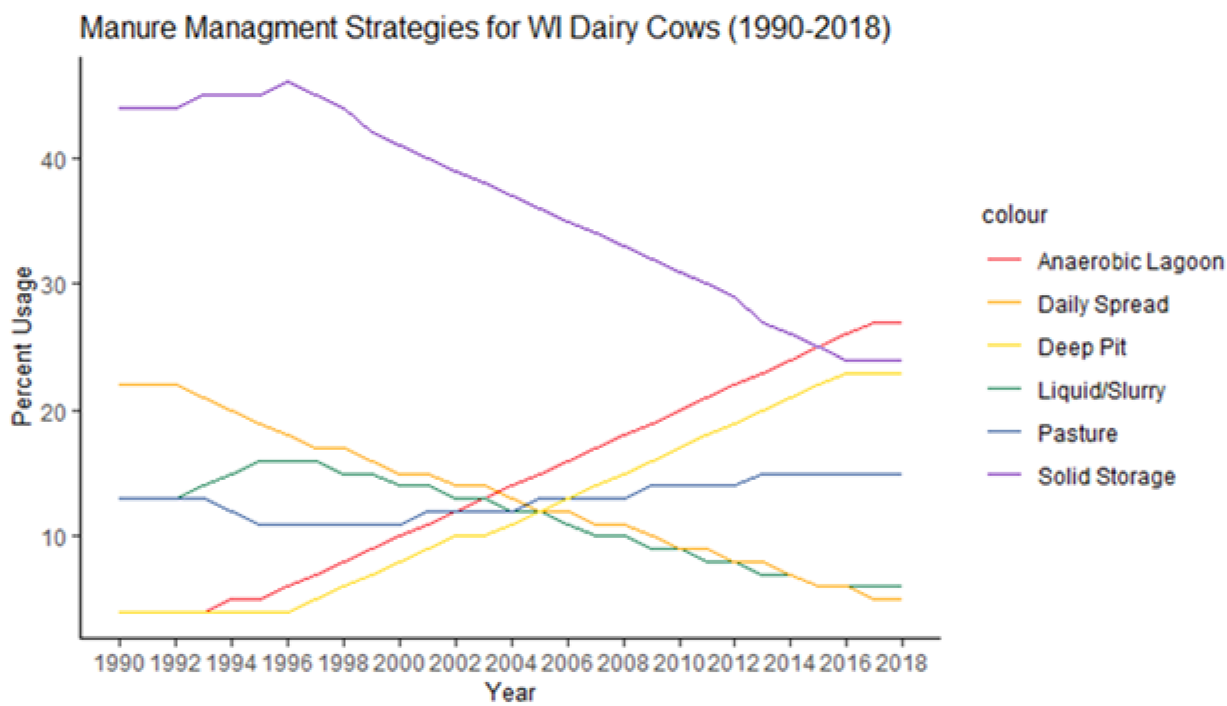

Figure 7. Percent of dairy cow manure managed by various practices in Wisconsin over time, as estimated by the EPA in the State Inventory Tool.

As a result of this shift in manure management, the state weighted methane conversion factor (i.e., the sum of the proportion of manure in the state managed by a practice multiplied by that practice's methane conversion factor) has increased 3-fold from 7% in 1990 to 23% in 2018 (Fig. 8). As discussed above, methane accounts for the majority of GHG emissions from manure management resulting in the direct relationship between MCF increase and total emissions increase.

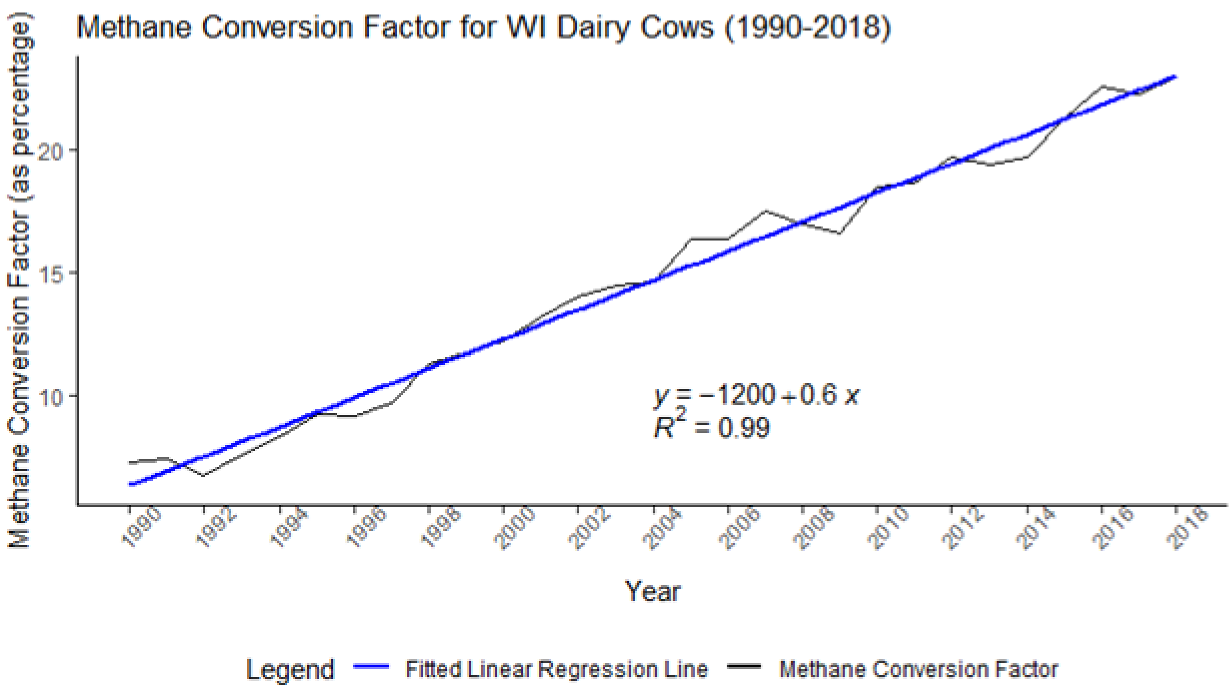

Figure 8. Increase in overall state-weighted methane conversion factor for dairy cow manure management in Wisconsin, as estimated by the EPA in the State Inventory Tool.

Strategies to reduce emissions from manure management include:

- Reducing storage time through increased daily spread or pasturing.
- Composting to increase solid manure management in aerobic conditions to reduce methane production

- Solid-liquid separation moves volatile solids into dry, aerobic storage conditions, reducing methane production. Mechanical separation can separate 45% of the solids from the manure.[3]
- Covering liquid storage allows for the capture and destruction of methane through flaring.
- Anaerobic digesters capture and destroy or use methane.

The scope of this analysis does not include the following:

- Potential feed additives to reduce enteric emissions. Enteric emissions represent a significant amount of GHG emissions from the agricultural sector in WI. There is considerable interest in developing feed additives/supplements to reduce these emissions, and some are promising, such as 3-NOP with meta-analyses indicating over 30% reductions in enteric emissions (Dijkstra et al. 2018; Kebreab et al. 2023). However, studies to date are short-term (up to several months), and the long-term efficacy of supplements in reducing enteric emissions are highly uncertain. Indeed, some of the longer-term studies indicate that emissions begin to return to baseline levels as the rumen microbial community adjusts to the supplement (Melgar et al. 2020, 2021; Schilde et al. 2021).

- Electricity/fuel usage on the farm itself, as this is not included in the agricultural sector of WDNR GHG inventory we are using as our baseline.

- Soil carbon flux beyond the sequestration potential of a conversion from an annual system to perennial systems that we credit towards offsetting agricultural sector emissions as discussed above.
- Potential societal diet changes to reduce demand for animal products
- Land-use conversion to/from agricultural land

Note that while we do not consider these as primary options, we do briefly explore what reduction in milk demand or enteric emissions from the dairy industry would be needed to close

[3] https://learningstore.extension.wisc.edu/products/solid-liquid-separation-of-manure-and-effects-on-greenhouse-gas-and-ammonia-emissions-p1844

the gap between our most optimistic NCS adoption scenario and reaching net-zero in the agricultural sector.

4. LIMITATIONS

This is a first-of-its-kind, high-level analysis exploring the challenge of net-zero agriculture in Wisconsin, establishing a foundation from which future analyses can build and improve. In particular, the agricultural module in the SIT is incomplete and this analysis is effect only looking at the potential to offset current livestock, fertilizer, and crop residue emissions. Future analyses should develop a more comprehensive baseline inventory of agricultural emissions that adds fuel and electricity consumption and existing soil carbon flux to the existing components of the EPA SIT agricultural module.

It is important to note that the true, realized climate mitigation result of practice implementation on a given farm can be highly variable. For example, soil carbon sequestration of a given practice is dependent on a number of factors including the specifics of how the practice is implemented (e.g., species used, timing and duration of implementation), cropping system, prior field management, local climate, and field soil characteristics. This means that even within a given farm, the climate change mitigation potential of implementing a given practice can vary substantially from field to field.

Thus, these estimates are best interpreted as high-level estimates of the relative climate change mitigation potential for Wisconsin agriculture, rather than precise, absolute predictions of what will happen in Wisconsin if these practices are implemented.

5. GREENHOUSE GAS MITIGATION POTENTIAL RATE SELECTION

5.1 General Comments on Soil Carbon Sequestration

First a general note with respect to soil carbon sequestration applicable to all practices that rely at least in part on soil carbon sequestration. Long-term research from the University of Wisconsin's Arlington Research Station has raised concerns that existing estimates may be overestimating the soil carbon sequestration potential of agricultural practices (Deitz et al. 2024, and citations therein). This work has identified some important shortcomings of many of the existing analyses that could be causing overestimates of the soil carbon sequestration.

First, rather than using long-term monitoring following the implementation of a practice on a field to evaluate carbon stock change due to the practice, many studies compare concurrent soil carbon stocks in fields with a practice implemented on it to stocks in reference fields without the practice implemented and assume that stocks in reference fields remain constant. Thus, in this "space-for-time" substitute approach, any increase in carbon stocks in the managed field compared to the reference fields are considered to reflect soil carbon sequestration. However, if reference fields are losing carbon rather than remaining constant, the interpretation of any observed increased carbon in the managed fields relative to the reference field is quite different. At a minimum it would reduce any carbon sequestration benefit from the management practices but the relative increase may only represent slower loss or maintenance of carbon, rather than carbon accrual that could offset emissions elsewhere. To overcome the "space-for-time" limitation, long-term monitoring from fields with practices implemented on them are needed to evaluate whether soil carbon is actually increasing.

Second, many studies only measure soil carbon in the top 15 or 30 cm of the soil profile. However, gains in the surface soil may be partially or even fully offset by losses deeper in the soil profile. In the case of a partial offset, the climate benefit of a practice only evaluated in the surface soil will be overestimated. In the case of a full offset, the practice will not be providing any net soil carbon increase. To overcome this limitation, sampling to deeper depths (e.g., at least 60 cm; Raffeld et al. 2023) is needed.

Finally, changes in soil bulk density that often accompany management changes are not always accounted for. A change in bulk density will change the mass of soil sampled at a given depth. For example, if adopting no-till increases soil bulk density, the mass of soil from 0-30 cm at the start of adoption will be more than that of the same profile after years of no-till adoption. However, if this bulk density change is not accounted for, the change in soil carbon stock will be overestimated.[4] To overcome this limitation of fixed depth sampling, using an equivalent soil mass approach is recommended (Raffert et al. 2024).

Indeed, comprehensive data (long-term monitoring up to 90 cm soil depth) from Arlington Research station convincingly demonstrate these limitations. The comprehensive dataset shows that the reference fields, fields with cover crops and no till practices implemented, and semi-perennial fields (corn-alfalfa rotations) have all lost soil carbon over the past 30 years, while rotationally-grazed pasture and restored prairie have maintained their soil carbon (Dietz et al. 2024). However, using incomplete methodologies ("space-for-time" substitution, shallow sampling, fixed depth sampling) on this dataset resulted in overestimations of soil carbon increases, including the reference fields, cover cropped fields, and semi-perennial fields maintaining carbon and the pasture and prairie fields gaining carbon (Dietz et al. 2024).

Given these findings and the prevalence of these limitations in the studies underlying existing estimate of the soil carbon sequestration potential of these practices, we set the lower soil carbon sequestration potential for all field-management practices to zero.

However, we also note that another long-term study from Michigan that overcomes the common limitations highlighted by Dietz et al. (2024) found that soil carbon was maintained in conventional agricultural fields and that some conservation practices have resulted in a soil carbon gains over 25 years (Córdova et al. 2025). This illustrates the site- and management-specific nature of soil carbon sequestration dynamics, and the potential of some practices to result in carbon sequestration not observed at the Arlington Research Station.

It is also important to note some temporal aspects of soil carbon sequestration in agricultural fields. First, any carbon gains are only in place as long as the practice is maintained. If the practice is discontinued, any stored carbon is likely to be released back into the atmosphere. This underscores the importance of irreversible reductions like reduced livestock emissions and fertilizer use reductions. Second, the soil carbon sequestration rates will slow and then stop as the soil reaches carbon saturation. For this analysis, we are relying on Fargione et al. (2018) assumptions that time

[4] Similarly, if a practice reduces soil bulk density, failing to account for the change will underestimate the change in soil carbon stock.

to saturation for these practices is greater than 50 years. However, to the extent that saturation times are significantly lower, our analysis results should be considered optimistic.

5.2 No-till soil carbon sequestration flux

We identified eight potential soil carbon sequestration rates for adoption of no-till ranging from 0-0.80 Mg CO_2 ac^{-1} yr^{-1} (Table A.1).

Early interest in no-till as a mechanism for increasing soil carbon was based on the following pieces of evidence: 1) the well-understood relationship between conversion to cropland, which is cultivated by plowing, and large soil organic carbon (SOC) losses; 2) research showing that no-till increases and stabilizes soil aggregates, protecting carbon from microbial decomposition; and 3) empirical studies documenting more carbon in soils that had been converted to no-till (Ogle et al. 2019).

However, early studies only looked at the top 30 cm of soil which have the most gains in SOC because no till leaves organic material on top of the soil and doesn't redistribute SOC deeper into the soil. However, subsequent studies analyzing deeper into the soil profile found that carbon gains at the top from no-till are offset by reduced carbon deeper in the soil (e.g., Luo et al. 2010, Powlson et al. 2014, Haddaway et al. 2018).

Furthermore, studies often fail to correct for increased soil bulk density in no-till soils, leading to overestimates of carbon sequestration under no-till (Powlson et al. 2014) and there is evidence that no-till can increase N_2O emissions (e.g., Six et al. 2004, Guenet et al. 2021) offsetting any carbon storage benefit.

Specific to cool moist climates like Wisconsin, specifically, there is less certainty for any overall SOC gains from no-till, particularly in silty/loamy/clayey soils (Ogle et al. 2019). One of the mechanisms by which no-till can increase SOC is through protection of SOC by reducing disturbance and increasing aggregate stability. However, in colder climates, there are limits to the protection no-till can provide due to disturbance from freeze-thaw cycles.

Taken collectively, the early promise of no-till to mitigate climate change through carbon sequestration is greatly undermined. Indeed, a recent review concluded that no-till should be best viewed as a means of adapting to climate change through improved soil health rather than as a tool to mitigate climate change through carbon sequestration (Ogle et al. 2019). Indeed, two quantifications of natural climate solution potential globally (Griscom et al. 2017) and in the United States (Fargione et al. 2018) chose not to include no-till as a mechanism with sufficient confidence in its efficacy.

The most robust Wisconsin-specific information is available from the long-term soil studies at the Arlington Agricultural Research Station, where long-term studies have not found no-till to sequester atmospheric carbon (Sanford et al. 2012; Rui et al. 2022, Dietz et al. 2024).

For these reasons, we use a value of 0.03 Mg CO_2eq ac^{-1} yr^{-1} as the upper end of the best estimate range for no-till climate mitigation in Wisconsin. This is the estimate for Eastern Canada used by Drever et al. (2021) and reflects the limited certainty that no-till will actually result in carbon accrual but still acknowledging that some studies do indeed find a soil carbon benefit.

5.3 Cover crop soil carbon sequestration flux

We identified nine potential soil carbon sequestration rates for adoption of cover crops ranging from 0.18-1.09 Mg CO_2 ac^{-1} yr^{-1} (Table A.2). The sequestration rate from Poeplau and Don (2015) of 0.47 Mg CO_2 ac^{-1} yr^{-1} is frequently used in prior quantifications of NCS potential on agricultural land at national or global scales. However, Poeplau & Don's value is based on a collection of global studies, and the current understanding of cover crop carbon sequestration in Wisconsin suggests that the potential sequestration is lower than a global estimate.

The effectiveness of cover crops in sequestering carbon is highly dependent on local conditions and specific implementation (e.g., leguminous vs. non-leguminous species; timing of planting). For example, Blanco-Canqui (2022) found that cover crops only increased carbon stocks in 1/3 of comparison studies in the United States. One of the most important factors determining the soil carbon sequestration potential of cover crop adoption is the amount of biomass produced by the cover crop (McClelland et al. 2021; Blanco-Canqui 2022; Wooliver & Jagadamma 2023, Joshi et al. 2023). In cooler, higher latitude fields, cover crops that are planted following harvest of the primary cash crop do not have many growing degree days to accumulate much biomass and develop extensive root structure, limiting the sequestration potential of cover crops in such environments. Indeed, McClelland et al. (2021) found that the 95% confidence interval of soil carbon sequestration in temperate, cool agroecological zones (which encompasses Wisconsin) include no increase in soil carbon, and Jian et al. (2020) found a negative relationship between SOC gains from cover cropping and both latitude and mean annual temperature.

Specific to Wisconsin, long-term studies from the Arlington Agricultural Research Station had found that cover cropped fields were not sequestering carbon over the past 30 years when the full soil profile is considered (Sanford et al. 2012, Cates & Jackson 2018; Cates et al. 2018, 2019; Rui et al. 2022, Dietz et al. 2024). As with no-till practices, as more research explores the soil carbon effects of cover crops at depth, there is increasing awareness that any gains in the surface soil can be offset by losses in deeper soils (McClelland et al 2021, Dietz et al. 2024).

For these reasons, we use the value of 0.18 Mg CO_2eq ac^{-1} yr^{-1} from Blanco-Canqui (2022) as the upper end of our best estimate range for cover crop climate mitigation in Wisconsin. This is on the lower end of reported sequestration values (which, as described above, we could expect for Wisconsin if there is any sequestration), is specific to the United States with good representation from the midwestern United States, and is within the range of estimates from the COMET-Planner model.

5.4 Conversion of Annual Row Crop to Perennial Herbaceous Crops

The soil carbon benefit from meta-analyses analyzing the conversion from annual crops to perennial herbaceous crops are summarized in Table A.3. These potentials are derived nearly entirely from bioenergy grasses (switchgrass, *Miscanthus*) and alfalfa. An additional meta-analysis evaluated soil carbon changes in fields following a conversion to perennial crops[5] (largely bioenergy crops and forage crops) but did not report the rate of change, but rather a percent difference (Siddique et al. 2023). Consistent with the analyses in Table A.3, they found perennialization increased soil carbon, reporting an increase of 17-23% in the top 30 cm soil (Siddique et al. 2023).

We are unaware of any literature reviews or meta-analyses of the potential for Kernza® to increase soil carbon stocks compared to annual crops. However, van der Pol et al. (2022) sampled three sites in Kansas that had been converted from annual grains to Kernza® between 5 and17 years prior and found that the fields accrued SOC at a rate of 0.61 Mg CO_2eq ac^{-1} yr^{-1} across 0-100 cm

[5] Note that this analysis is not strictly related to herbaceous perennials since ~20% of the studies included in the analysis included woody perennials (e.g., poplar, willow).

soil depth compared to annual crop fields. This is consistent with some of the longer-term studies of other perennial herbaceous crops summarized in Table A.3.

Specific to Wisconsin, as noted above, long-term research at the Arlington Research Station found semi-perennialization (alfalfa and corn rotation) was found to continue to lose carbon (although at a slower rate) compared to a continuous corn rotation while full perennialization in the form of a grass-based pasture or prairie maintained carbon (i.e., neither gained or lost carbon).

For the upper end of our best estimate range for the conversion of annual crops to perennial herbaceous crops, we use 1.26 Mg CO_2eq ac^{-1} yr^{-1}, which is the average of the available meta-analyses.

5.5 Conversion of Annual Row Crop to Grassland or Well-managed Rotationally-Grazed Pasture

In the most comprehensive meta-analysis we are aware of, Conant et al. (2017) found that conversion from cropland to grassland increased soil carbon by 1.30 Mg CO_2eq ac^{-1} yr^{-1}. This dataset included 93 studies with a global scope, but a strong bias toward temperate North America, and had a mean sample depth of 44.5 cm.

Fargione et al. (2018) use a sequestration flux of 1.78 Mg CO_2eq ac^{-1} yr^{-1} for restoring cropland to grassland in the United States, and this average value is representative of estimates for Wisconsin specifically (see Figure S19 in Fargione et al. 2018).

Kaempf et al. (2016) found that the average sequestration rate in temperate grassland following conversion from cropland was 1.07 Mg CO_2eq ac^{-1} yr^{-1}

Specific to Wisconsin, Becker et al. (2022) compared soil carbon in paired pastures and row crops in central and southern Wisconsin. They report that pastures had significantly more surface (0-15 cm) carbon than row crop counterparts and that soil carbon increased with pasture age at a rate of 0.49 Mg CO_2eq ac^{-1} yr^{-1}. Although this increase in soil carbon with pasture age does not directly reflect a conversion from agricultural fields, we assume that soil carbon accumulation rates in row crop agriculture are likely close to zero, if not negative and thus this value represents a conservative potential increase in soil carbon that could be seen under a conversion to pasture.

At the long-term Arlington Research Station, Rui et al. (2022) found that after 29 years pasture fields had 18-29% more soil carbon in the top 30 cm than any of the annual crop fields and conclude that grazed perennial grasslands have the potential to accumulate soil carbon in Wisconsin's grassland soils. However, Sanford et al. (2022) and Dietz et al. (2024) report that soil carbon gains in the top 30 cm can be offset by losses deeper in the soil profile, resulting in pasture maintaining its original carbon (i.e, not losing or sequestering carbon).

Finally, in Michigan, Stanley et al (2018) found that adaptive multi-paddock grazed pastures sequestered carbon at a rate of 5.34 Mg CO_2eq ac^{-1} yr^{-1} in the top 30 cm of soil.

We note that, as with other practices to increase soil carbon, the potential of grasslands and pastures to sequester carbon is site- and context-specific. Soil type and initial conditions, as well as local environmental conditions will strongly influence the amount of carbon storage. Management also plays an important role; for example, intensity and type of grazing and incorporation of legumes have been shown to affect soil carbon storage rates (e.g., Oates and Jackson 2014, Conant et al. 2017).

We use the 1.30 Mg CO_2eq ac^{-1} yr^{-1} from Conant et al. (2017) as the upper end of our best estimate range as it reflects the most statistical power coming from 93 different studies, and lies in between more recently-published individual field data points in the Upper Midwest.

5.6 Improved Grazing Management

Meta-analyses have found that rotational grazing increases soil carbon compared to continuous grazing (Byrnes et al. 2018) or that lighter grazing increases soil carbon sequestration in pastures (McSherry & Ritchie 2013; Zhou et al. 2017).

Fargione et al. (2018) used a national value of 0.07 Mg CO_2eq ac^{-1} yr^{-1} in soil carbon sequestration due to grazing optimization from a dataset compiled by Henderson et al. (2015). However, this dataset reports no expected sequestration in WI (see Fig. S17 in Fargione et al. 2018).

A synthesis by Conant et al. (2017) reports a global potential of 0.42 Mg CO_2eq ac^{-1} yr^{-1} for improved grazing. Similarly, a review from Smith et al. (2008) reports a global potential in humid climates of 0.32 Mg CO_2eq ac^{-1} yr^{-1} from improved grazing practices.

As with other practices, the soil carbon sequestration potential for optimized grazing intensity will vary by specific implementation: geography, soil characteristics, baseline soil carbon stocks, and historic pasture management (Godde et al. 2020). Based on these values we use a range of 0-0.42 Mg CO_2eq ac^{-1} yr^{-1} for improved grazing management.

5.7 Agroforestry carbon sequestration fluxes

The carbon sequestration of a given practice is highly dependent on the specific implementation, including species, age, tree density, agroecological conditions and soil conditions (Feliciano et al. 2018). In general, there are fewer studies of agroforestry carbon impacts in cool, temperate areas than there are for other practices like no-till or carbon, and none from Wisconsin that we are aware of. Most agroforestry carbon sequestration studies are from tropical areas, limiting their applicability to Wisconsin, so to the extent that reviews and meta-analyses break down their results by agroecological zone we extracted values most relevant to Wisconsin.

We also note that the relatively small number of studies examining the carbon dynamics of transitioning from annual systems to agroforestry systems leaves the possibility that further study will find less potential, along the lines of how increased study of cover crops and no-till practices found reduced potential after initial optimism. However, most of the reported sequestration potential from agroforestry systems comes from tree biomass, which is significantly less uncertain than soil carbon sequestration. Additional study may find less optimistic soil carbon gains from agroforestry systems, but there will always be some sequestration from the biomass carbon accumulation.

5.7.1 Agroforestry carbon sequestration fluxes: alley cropping

Studies presenting alley cropping carbon sequestration fluxes are summarized in Table A.4. Reported fluxes ranged from 1.30 to 5.06 Mg CO_2eq ac^{-1} yr^{-1}. As an upper estimate we use 2.19 Mg CO_2eq ac^{1} yr^{-1}, from Fargione et al. (2018), and which is also similar to the global value from Feliciano et al, and all temperate, cool studies from Cardinael et al. (2018). For a lower estimate we use 1.30 Mg CO_2eq ac^{-1} yr^{-1}, from Drever et al. (2021), the lowest reported value.

5.7.2 Agroforestry carbon sequestration fluxes: windbreaks

There are limited reported carbon sequestration fluxes from windbreaks (Table A.5). For our best estimate range we use this full range of all reported values (1.42 to 5.26 Mg CO_2eq ac^{-1} yr^{-1}) as no individual reported values stood out as being more robust or applicable to Wisconsin than others.

5.7.2 Agroforestry carbon sequestration fluxes: Silvopasture

Carbon sequestration fluxes from silvopasture are summarized in Table A.6. As an upper estimate we use 2.36 Mg CO_2eq ac^{-1} yr^{-1}, from Feliciano et al.'s (2018) North American value, which is also similar to Cardinael et al.'s (2018) cool/temperate North American values. For a lower estimate we use 1.23 Mg CO_2eq ac^{-1} yr^{-1}, from Drever et al. (2021), the lowest reported value.

5.7.3 Agroforestry carbon sequestration fluxes: Riparian Buffers

Carbon sequestration fluxes from riparian buffers are summarized in Table A.7. As an upper estimate we use 6.68 Mg CO2eq ac^{-1} yr^{-1}, from COMET Planner estimates in Wisconsin, which is the highest reported value. For a lower estimate we use 3.74 Mg CO_2eq ac^{-1} yr^{-1}, from Drever et al. (2021) the lowest reported value other than Kim et al., (2016), which only included a single study site.

5.8 Nitrogen Fertilizer Use Changes Due to Conversion from Annual Row Crops

We also accounted for changes in nitrogen fertilizer use due to conversion from corn or soybeans to the NCS practices discussed here. The precise changes will depend on the local soil conditions and the particulars of the species being established, but Table 1 summarizes the general changes in fertilizer use that we assumed in our calculations.

Table 1. Nitrogen fertilizer changes as a result of annual row crop conversion.			
Conversion	**Prior N fertilizer application rate (pounds per acre) for corn/soy**[1]	**NCS N fertilizer application rate (pounds per acre)**	**Notes**
Corn/soy to perennial herbaceous crop	180/5	60	Based on 80 pounds per acre for switchgrass, miscanthus, Kernza and 5 pounds per acre for alfalfa (Laboski & Peters 2012, Pennington 2012, Tautges et al. 2023)
Corn/soy to pasture	180/5	0	Assuming pasture fertilized only by manure deposits from grazing cows (E.g., Jackson 2022)
Corn/soy to forested riparian buffer	180/5	0	Assume no fertilizer applied to forested buffer
Corn/soy to windbreak	180/5	0	Assume no fertilizer applied to windbreaks
Corn/soy to alley crop	180/5	50	Based on 50-100 pounds per acre for poplar; 20-100 for nut trees, including chestnut and hazelnut; 50 pounds per acre for fruit trees; and 30 pounds per acre for berries (Braun n.d., McDonald n.d.,University of Georgia n.d., McLaughlin et al. 1987, Cheng 2010, Laboski & Peters 2012, Lizotte & Mandujano 2015, Buchman et al. 2020, Lowenstein & Crain 2025,)
[1] Based on Laboski & Peters 2012			

5.9 Biochar Soil Amendments

The climate mitigation potential of biochar amendments follows IPCC methodology and is calculated from Woolf et al. (2021) as:

$$\begin{aligned} &Mitigation\ potential \\ &\quad = Mass\ biochar\ \times organic\ carbon\ content\ of\ biochar \\ &\quad \times fraction\ biochar\ carbon\ remaining\ after\ 100\ years \end{aligned}$$

Fargione et al. (2018) and Drever et al. (2021) used agricultural residues as biochar feedstock in their analyses; however, the NRCS does not recommend using agricultural residues as a feedstock since leaving it place provides important soil protection and soil health benefits. Thus, we use woody biomass from forestry residue that can be economically and sustainably removed (i.e., leaving residues important for ecological services in place). Springer et al. (2017) estimate that Wisconsin sustainably produces between 1 and 2 million dry tons of forestry residue annually. Assuming that one ton of woody biomass feedstock produces 0.42 tons of biochar (Woolf et al. 2010), this represents an annual potential of 420,000-840,000 tons of biochar. Biochar can be applied to the plow layer at a rate of 50 tons per hectare per 100 years (Woolf et al. 2010). Given the amount of cropland in the state, this annual supply of biochar corresponds to 278-555 years of applications, meaning that we have more than enough agricultural land to be incorporating this biochar through at least 2050.

To calculate the greenhouse gas mitigation potential of this annual biochar application, we assumed an organic carbon content of the biochar of 76% and that 85% of the carbon remains after 100 years (Woolf et al. 2021). This results in 1-2 MMT CO_2eq of biochar greenhouse gas mitigation potential on agricultural lands.

We note that our sole focus on persistence-derived carbon sequestration from binding the carbon up in the biochar may under- or overestimate net total lifecycle greenhouse gas balances from biochar applications. This sequestration is typically the largest impact on net GHG balances (Woolf et al. 2021), but future analyses could include other GHG impacts for a more complete analysis. For example, on the one hand, there are transportation and production emissions associated with biochar. On the other hand, biochar applications can also reduce soil N_2O emissions and reduce nitrogen fertilizer needs by improving soil fertility. Production emissions can also be minimized through co-production of bioenergy or the use of renewable energy sources (Woolf et al. 2021). Lacking a standardized methodology for a full lifecycle analysis, we are following the methodology of prior analysis from Fargione et al (2018) and Drever et al. (2021) and include only the persistence-derived carbon sequestration.

5.10 Nitrogen Fertilizer Management

The climate change mitigation potential from improved nitrogen management is calculated as the avoided GHG emissions associated with nitrogen fertilizer production and the reduction in soil N_2O emissions from nitrogen inputs to fields.

Fargione et al. (2018) provide upstream emissions associated with each type of nitrogen fertilizer (e.g., anhydrous ammonia, urea, etc.). Based on current use of each type in the U.S., they provided an overall estimate of 4.41 g CO_{2eq} per gram of N fertilizer in the United States, which is what we use in this report. This is consistent with other estimates of 3.9 g CO_2eq per g N fertilizer from Carmago et al. (2013), an estimate of 4.05 g CO_2eq per g N fertilizer in Canada from Dyer et al. (2017), and a global range of 3.2-6.6 g CO_2eq per g N fertilizer from Bellarby et al. (2008).

N_2O emission quantification is done using emissions factors, which relate N_2O emissions to total N-input. Emissions factors represent the percent of total N input that is subsequently emitted as N_2O; for example, an emissions factor of 1% indicates that 1% of all the N applied to a field will be emitted to the atmosphere as N_2O.

There are two approaches to calculating N_2O emissions factors: bottom-up and top-down. Bottom-up approaches divide emissions between direct emissions (i.e., directly from the cropland) and indirect emissions (N_2O emissions from ecosystems downstream/downwind of agricultural land which receive reactive nitrogen from leaching, run-off or atmospheric redeposition), using different emissions factors for direct and indirect emissions. Top-down approaches relate changes in total measured atmospheric N_2O concentrations to changes in N-inputs, accounting for changes in industrial emissions and emissions from non-agricultural land (Smith et al. 2017).

There is some broad convergence between top-down and bottom-up estimates at a global scale (Del Grosso et al. 2008, Smith et al. 2012). However, some regional analyses have found that bottom-up approaches underestimate N_2O emissions compared to top-down approaches, potentially due to hidden "hot-spots" of N_2O emissions within a landscape and errors in the indirect emissions factors, which are relatively understudied (Griffis et al. 2013, Turner et al. 2015). For example, a study in southern Minnesota found that measured N_2O emissions from streams are up to 9 times greater than commonly used indirect emissions factors, resulting in the bottom-up approach underestimating N_2O emissions by 40%. (Turner et al. 2015). Specifically, zero-order streams (e.g., headwaters, drainage ditches) enriched with reactive nitrogen from runoff and leaching from agricultural land are N_2O emission hotspots that can double agricultural emissions when appropriately accounted for.

Direct soil emission factors are the most studied and estimates relevant to Wisconsin are summarized in Table A.8. Top-down emissions factors reported in the literature range from 2-5% (Table A.9).

We follow the EPA's Greenhouse Gas Emissions and Sinks methodology[6] for quantifying greenhouse emissions from synthetic fertilizer applications in Wisconsin. This is the methodology used by the WDNR in its Greenhouse Gas Inventory, which is the baseline we use for this Roadmap. The emissions factors used in this approach are summarized in Table A.10.

Overall, this approach results in a total emissions factor of 1.2 kg N_2O-N per kg N synthetic fertilizer. This is lower than top-down emissions factors, such as the 2.3% from Thompson et al. 2019, which was used in the NCS potential calculations for the United States used by Fargione et al. (2017). This is also lower than emission factors from the 2019 IPCC revisions (summarized in Table A.10), which would result in 1.84% for wet climates. Thus, this should be considered a conservative potential for the climate mitigation potential of synthetic N fertilizer management. This is particularly true when considering the non-linear relationship between N_2O emissions and nitrogen fertilizer application rates (e.g. Hoben et al. 2010). Reductions at the higher end of the rate scale will have disproportionately high reductions in N_2O emissions that aren't reflected in the SIT methodology.

The total mitigation potential for reductions in nitrogen applications to fields is calculated as:

$$\begin{gathered} N\ Fert.\ Management\ Potential = \\ (\ ProductionEmissions_{Fert,\ current} + N_2OEmissions_{Fert,\ current}) \\ -\ (\ ProductionEmissions_{Fert,red} + N_2OEmissions_{Fert,red}) \end{gathered}$$

where $Fert,_{current}$ is for current nitrogen application and $Fert,_{red}$ is reduced nitrogen fertilizer application.

5.11 Manure Management

The general equations for quantifying the GHG emission from manure management are:

$$\begin{gathered} Methane\ emissions = total\ volatile\ solids\ in\ manure \times methane\ conversion\ factor \\ N_2O\ emissions = total\ nitrogen\ in\ manure \times N_2O\ emission\ factor \end{gathered}$$

[6] https://www.epa.gov/ghgemissions/methodology-report-inventory-us-greenhouse-gas-emissions-and-sinks-state-1990-2021

The methane conversion factor and N_2O-N emission factor depend on climate and manure form and management system (Table A.11).

We follow the EPA's Greenhouse Gas Emissions and Sinks methodology[7] for quantifying greenhouse emissions from manure management in Wisconsin. This is the methodology used by the WDNR in its Greenhouse Gas Inventory, which is the baseline we use for this Roadmap.

To calculate 2018 methane emissions, the EPA calculated a Wisconsin-specific methane conversion factor of 23% for dairy cows. This is based on apportioning total statewide manure management among six practices: pasture, daily spread, solid storage, liquid slurry, deep pit, anaerobic lagoon, and anaerobic digesters. Based on the percentage of manure volatile solids handled by each practice and that practice's MCF, an overall state-weighed MCF of 23% is calculated (Table A.12).

For direct N_2O emissions, total nitrogen excretion is apportioned between liquid systems (anaerobic lagoons and liquid/slurry) and solid storage and other systems (in WI, solid storage and deep pit). The total nitrogen in liquid systems is multiplied by a liquid systems emissions factor of 0.1%. The total nitrogen in solid systems is multiplied by an emissions factor of 2%.

Direct N_2O emissions for pastured cows are calculated by multiplying the N excreted by pastured cows by an emissions factor of 2%.

Direct emissions from landspreading of managed systems and daily spread is calculated by multiplying the total N excreted that is managed or daily spread, less the amount volatilized (assumed to be 20%), and multiplied by an emissions factor of 1.25%.

Indirect N_2O emissions from volatilization are calculated by multiplying total N excretion by the amount volatilized (20%) and a volatilization emissions factor of 1%.

Indirect N_2O emissions from runoff and leaching is calculated by multiplying total unvolatilized N excretion, by the amount of N assumed to leach (30%) and an emissions factor of 0.75%.

[7]https://www.epa.gov/ghgemissions/methodology-report-inventory-us-greenhouse-gas-emissions-and-sinks-state-1990-2021

Quantifying the methane mitigation potential of manure management practices to reduce emissions involves shifting the proportion of total state manure managed in each of the seven classes according to adoption scenarios and calculating a new state-weighted MCF.

To reduce GHG emissions, we developed scenarios of increased manure management via the following: solid manure management, increased use of anaerobic digesters, and covering lagoons coupled with flaring captured methane.

Based on these shifted proportions of manure handled by the different practices, a new state-weighted MCF is calculated, which in turn is used to calculate methane emissions under that scenario. The total climate mitigation potential of manure management adjustments is calculated as the difference in emissions from BAU manure handling and the emissions from the manure management scenario.

The climate benefits of reduced methane emissions via increased management of solid manure management in aerobic conditions will be somewhat offset by increased N_2O emissions, which are accounted for in our calculations, as described above. However, the methane emissions decrease more than N_2O emissions increase, resulting in a net climate benefit through improved manure management.

5.12 Transition from Confined Dairy Production to Grazed Dairy Production

Transitioning from confined milk production to grassfed milk production has numerous clearly established health and nutrition, economic (Dartt et al. 1999; Wiedenfeld et al. 2022; Winsten 2024), and ecological benefits when grazed livestock are managed well (Franzluebbers et al. 2012; Rotz et al. 2020, Jackson 2024). Ecological benefits include little to no soil loss (Vadas et al. 2015); little phosphorus loss to surface waters via runoff (Young et al. 2023); low nitrate loss to groundwaters via leaching (Jackson 2020); improved water interception, infiltration, and storage reducing flooding (Basche & DeLonge 2019; Bendorf et al. 2021); little use of anti-biotics reducing resistance risks, little use of pesticides reducing human health risks (Gerken et al. 2024) and impacts on pollinators and birds, better air quality reducing human health impacts (Hill et al. 2019), improved habitat for biodiversity (Temple 1999, Lyons et al. 2000, Undersander et al. 2000). Many of these outcomes are likely to improve agriculture's resilience in the face of climate change, making this a climate-smart practice. However, the focus of this analysis is solely on the greenhouse gas emissions consequences of this transition.

Numerous studies have reported the carbon intensity of either confined dairy production or grazed dairy production in Wisconsin or the upper Midwest. Unfortunately, there is no standardized approach for calculating the carbon intensity of milk production. As a result, different analyses use different models, assumptions, and system boundaries making comparison across studies intractable.

Thus, we are limited to using studies that used the same approach to compare the grazed and confined systems. We identified five such studies that calculate the carbon intensity of confined and grazed milk production in Wisconsin. None of these studies modeled 100% grassfed milk production. All grazed systems modeled in these analyses supplemented grass with grain (up to 60% of grazing season dry matter intake from pasture) and assumed a confined system diet in the non-grazing months.

Overall, two of the five analyses found that grazed systems have comparable or lower cradle-to-farm gate milk production carbon intensities as confined systems (Reinemann & Cabrera 2013, CIAS 2019), two analyses found grazed systems had comparable to slightly higher carbon intensities (Dutreuil et al. 2014, Aguirre-Villegas et al. 2017), and one study found increased grazing had substantially higher carbon intensity (Aguirre-Villegas et al. 2022). A summary of these analyses can be found in the appendix (Tables A.13-A.18).

This lack of consensus is consistent with global analyses that find that carbon intensity was associated more with specific management practices on a given farm than by general system type (Wattiaux et al. 2019). Generally, grazed systems have higher enteric emissions due to the lower digestibility of grass and lower milk production rates per cow, while confined systems have more machinery and chemical inputs and higher manure storage emissions, particularly when using liquid manure storage. Except for Dutreuil et al. (2014), none of these analyses consider the potential for soil carbon storage in the pastures of grazed systems.

Rotz et al. (2020) modeled the cradle-to-farm gate carbon intensity of different dairy production systems in Pennsylvania finding that 100% grassfed dairy production was the most carbon intensive system (1.46 kg CO_2eq per kg fat and protein corrected milk; FPCM), followed by confinement (1.28 kg CO_2eq per kg FPCM) and grazed with grain (1.15 kg CO_2eq per kg FPCM). This is the most relevant analysis we are aware of comparing the confined to 100% grassfed dairy production in the United States. Although this comparison does not reflect potential soil carbon sequestration following the conversion of cropland to pasture, the paper notes that the soil carbon sequestration potential of converting cropland to pasture could reduce the grassfed system's milk carbon intensity below that of a confined system.

It is important to note that most studies assess carbon intensity of dairy agroecosystems on a per unit milk basis, which always favors approaches that produce more milk. This productivist framing assumes milk is scarce and pushes us away from absolute reductions in greenhouse gas emissions and other environmental pollution rather than relative reductions, which could result in overall higher emissions.

We identified four transition scenarios: 1) maintain milk production constant, but shift 25% of total production to grassfed; 2) maintain milk production constant but shift 47%[8] of total production to grassfed; 3) maintain milk cow herd size constant and shift it all to grassfed; 4) limit milk production to what can be produced by grassfed cows on the land currently being used by dairy production.

For each transition scenario, we calculated the difference in cradle-to-farm gate greenhouse gas emissions for milk production under the current confined system paradigm (assuming 1.28 kg CO_2eq per kg FPCM for all milk production) and the transition scenario (assuming 1.28 kg CO_2eq per kg FPCM for any milk production that remains in confined systems and 1.46 kg CO_2eq per kg FPCM for any milk production that is shifted to 100% grassfed).

We converted milk production to FPCM milk production using ratios for grassfed and confined systems reported by Ranathunga & Wattiaux (2017) from national data.

The cradle-to-farm gate emissions include emissions from sources not included within the EPA State Inventory Tool agricultural module, such as electricity and on-farm fuel combustion. Approximately 10-20% of cradle-to-farm gate life cycle analysis emissions are from such sources for both confined and grazed systems (Reinemann & Cabrera 2013, O'Brien et al. 2014,

[8] 47% was chosen rather than 50% as it is the amount of production that can be shifted to grassfed on the 1.6 million acres of non-livestock feed corn and soy available after the lower end of our NCS practice adoption.

Cabrera & Dutreuil 2014, Aguirre-Villegas et al. 2017, CIAS 2019, Aguirre-Villegas et al. 2022; see Tables A13-A18). Thus, we assumed that 85% of the emissions difference is appropriately included within the scope of our analysis focused on the agriculture module. This is consistent with our treatment of other practices with GHG emission consequences outside of the agricultural module such as not including the effects of no-till on farm fuel use.

To account for soil carbon sequestration associated with the conversion of row crop fields to pasture to accommodate the transition to grassfed milk production, we use a potential sequestration range of 0 to 1.30 Mg CO_2eq ac^{-1} yr^{-1} as discussed above, for any annual row crops converted to pasture.

For corn/soy cropland converted to pasture that is not currently used for milk production, we also account for avoided nitrogen fertilizer applications (assuming 180 pounds per acre corn and 5 pounds per acre soybean) and crop residue emissions, as calculated in the SIT agricultural module.

Total reduction in greenhouse gas emissions for the transition to grazing scenarios is calculated as:

Total Reduction in GHG emissions = [area converted from annual crop to pasture x soil carbon sequestration rate] + [(current milk production carbon emissions – pasture transition scenario milk carbon emissions) x 0.85] + avoided GHG emissions from nitrogen fertilizer and crop residues on row crop conversion not currently used for dairy production.

The relevant variables and assumptions used in these calculations are summarized in Table 2.

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Table 2. Relevant variables and assumptions used in the transition to grazing scenarios.		
Variable	**Value**	**Source**
Current land used for dairy production in WI	796,592 ha	Jackson 2024
Current row crop land used for dairy production in WI	345,655 ha	Jackson 2024
Current milk production in WI	14.4 billion kg milk per year	Jackson 2024
Current herd size	1,270,000 milk cows	Jackson 2024
Milk yield per unit land area in confined system	18,125 kg milk per ha per year	Jackson 2024
Milk yield per animal in confined system	11,369 kg milk per milk cow per year	Jackson 2024
Milk yield per unit land area in 100% grazed system	7,846 kg milk per ha per year	Jackson 2024
Milk yield per animal in 100% grazed system	6,641 kg milk per milk cow per year	Jackson 2024
Milk carbon intensity in confined system	1.28 kg CO2eq per kg FPCM	Rotz et al. 2020
Milk carbon intensity in 100% grazed system	1.46 kg CO2eq per kg FPCM	Rotz et al. 2020
Ratio of confined milk production to FPCM	0.964	Ranathunga & Wattiaux 2017
Ratio of confined milk production to FPCM	0.953	Ranathunga & Wattiaux 2017

PART II. Natural Climate Solution Practice Adoption Scenario Development and Statewide Mitigation Potentials

1. *SCENARIO OVERVIEW*

For each practice, we have defined two adoption scenarios, an aggressive upper estimate of adoption and a more conservative adoption scenario. The scenarios are built progressively as follows (and summarized in Table 3 and Table A.19).

We began by modeling a set of scenarios with practices that can be adopted within the current dominant paradigm of intensive annual row cropping and confinement dairy production. Within this set of scenarios, we first looked at the GHG mitigation potential of only cover crops and no-till adoption. Next, we added reduced nitrogen fertilizer use and improved manure management. Finally, we added biochar soil amendments.

In our second set of scenarios, we looked at the GHG mitigation potential of a transition from annual row crops to perennial systems and the introduction of trees in existing pasture. First, we looked at the conversion to perennial crops and agroforestry systems, while assuming cover crops +no-till + nitrogen management + biochar amendments scenario on the remaining annual cropland. Next, we looked at improved manure management on top of the conversion to perennial systems. Finally, we explore various scenarios of transitions to 100% grassfed milk production.

Table 3. Summary of NCS adoption scenarios. CC = Cover crop adoption; NT = no till adoption. See Table A.19 for more specific inputs into each scenario.

Working within current dominant system of annual row crops and confined dairy production			
Scenario 1	**Scenario 2**	**Scenario 3**	**Scenario 4**
CC + NT	(Scenario 1) + N Fertilizer Management	(Scenario 2) + Manure Management	(Scenario 4) + Biochar + Improved Grazing
Transition to Perennial Agriculture (excluding transition to grassfed milk production)			
Scenario 5	**Scenario 6**	**Scenario 6+**	
Conversion to perennial systems + CC + NT + N + Biochar on all remaining cropland + Improved Grazing	(Scenario 5) + Manure Management	(Scenario 6) + Avoided enteric/manure emissions (via reducing dairy food waste by 50%) to reach net-zero	
Transition to grassfed milk production			
Scenario 7	**Scenario 8**	**Scenario 9**	
(Scenario 5) + Maintain current milk production but shift 25-50% of milk production to grassfed.	(Scenario 5) + Shift to 100% grassfed milk production while maintaining the current milk cow herd size	(Scenario 5) + Shift to 100% grassfed milk production only using land currently supporting dairy production	

2. *SUMMARY OF FIELD-BASED PRACTICE MITIGATION POTENTIAL*

The per-acre greenhouse gas mitigation potential of the field-based practices are summarized in Figures 9a,b. These potentials are multiplied across the area of adoption defined in the scenarios below to arrive at a total mitigation potential for the practice. The range of potential values we used for our upper and lower estimates for a given practice are outlined in the rectangles in Figures 9a,b. The justification for these ranges is detailed in Part I of this Appendix.

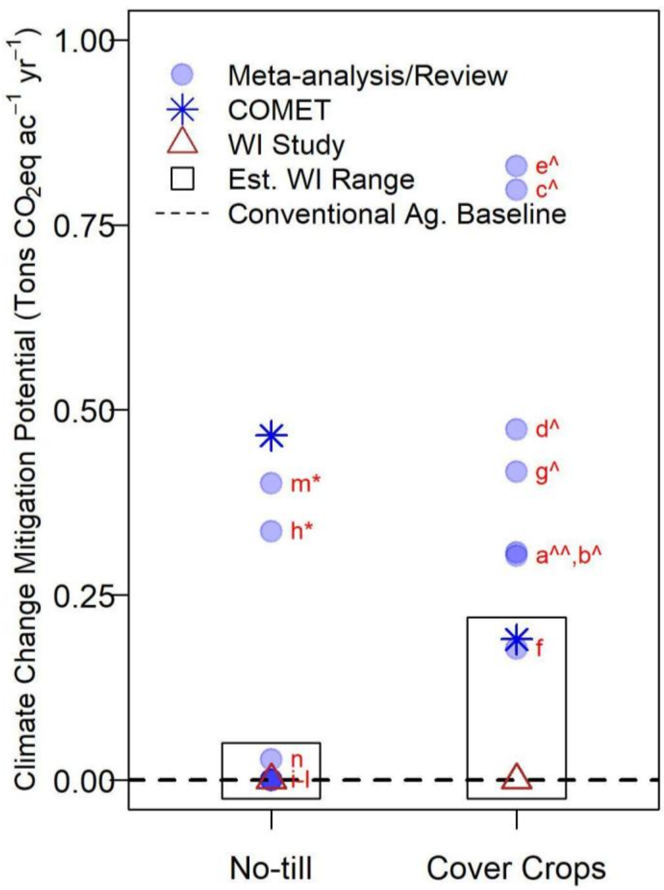

Figure 9a. Details of per-acre soil carbon sequestration of no-till and cover crops. In the no-till studies, *indicate studies that report sequestration only in the surface 30 cm of soil. The Wisconsin study site refers to findings from the Arlington Field Station (Dietz et al. 2024). In the cover crop studies, ^indicate global studies and ^^indicate temperate subsets of global studies. Study code: [a]McClelland et al; [b]King & Blesh; [c]Abdalla et al; [d]Poeplau & Don; [e]Jian et al; [f]Blanco-Canqui; [g]Joshi et al.; [h]Virto et al.; [i-l]Liang et al., Meurer et al., Haddaway et al, Luo et al.; [m]Ogle et al.; [n]Drever et al.

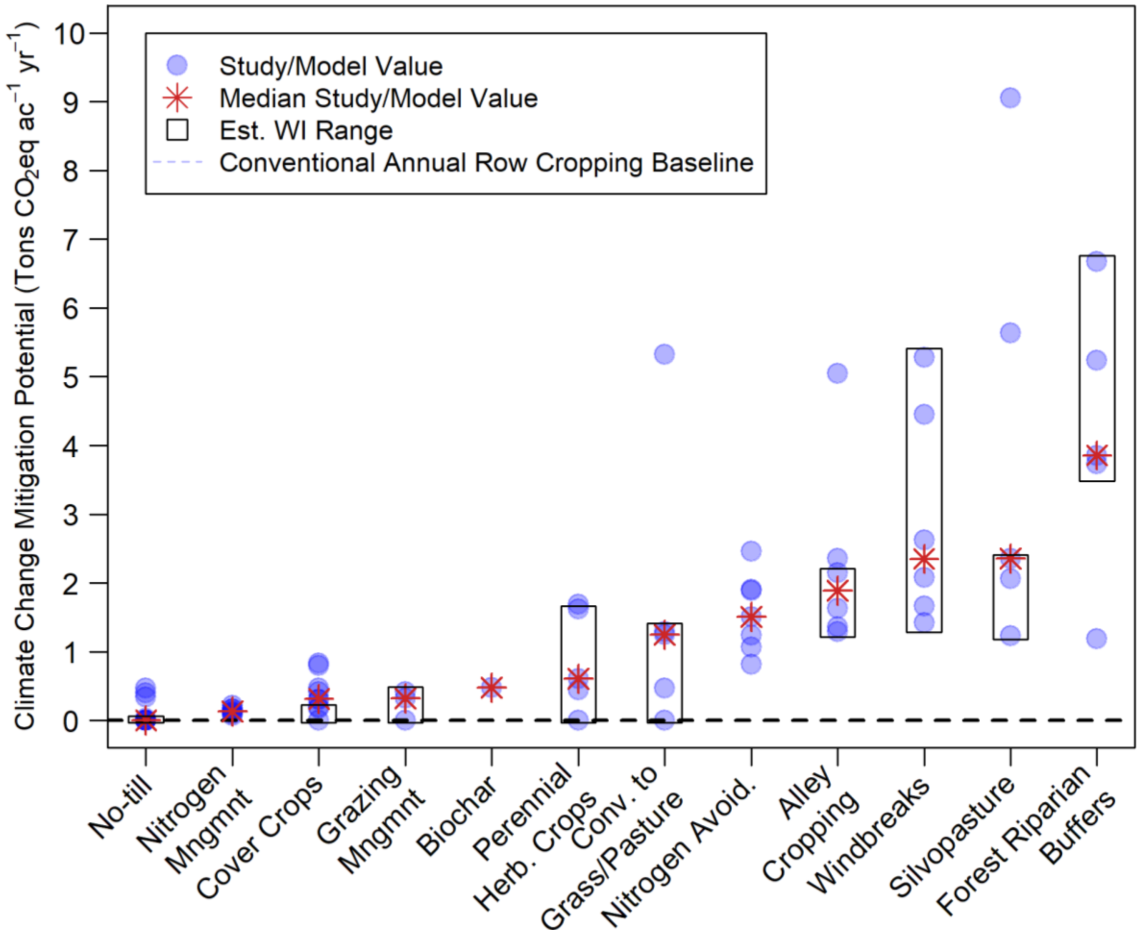

Figure 9b. Per-acre GHG mitigation potential of field-based practices, as reported in published literature for no-till and cover crops (left) and the full suite of field-based agriculture practices (right). *Nitrogen Management* values represent the N_2O reduction associated with a 20% reduction in nitrogen fertilizer use across all cropland statewide. Nitrogen Avoidance reflects conversion from corn (assuming 180 pounds N fertilizer per year; Laboski & Peters 2012) to a land use that does not use nitrogen fertilizer. The range of values within the boxes indicate the best estimates for Wisconsin that were used in our analysis. See Part I of this appendix for rationale behind the selected range of values.

3. *SCENARIOS WORKING WITHIN CURRENT DOMINANT ANNUAL ROW CROPPING PARADIGM*

3.1 Scenario 1: Cover crop and no-till adoption only

For the lower end of future adoption of cover crop and no-till, we extrapolate from trends in adoption in the USDA's Census of Agriculture from 2012-2022 (with 2012 being the first year that acreage of these practices is reported).

If these linear trends were to continue to 2050, 20% of non-forage cropland will have cover crops and 65% of non-forage cropland will have no-till adoption (Figure 10).

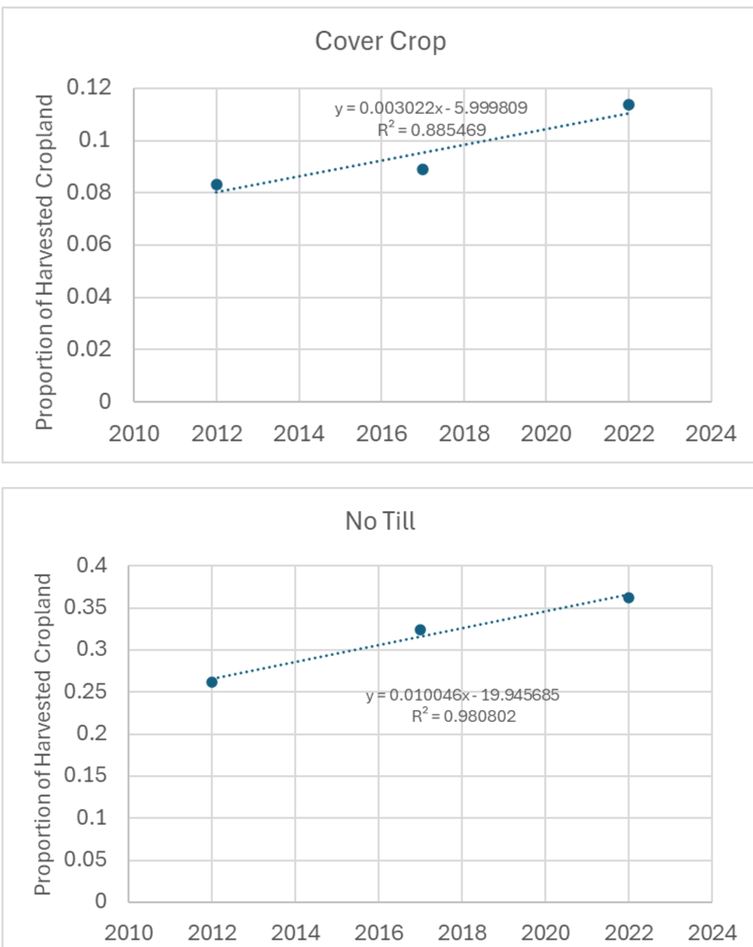

Figure 10. Linear trends in cover crop (top) and no-till (bottom) adoption on harvested cropland in Wisconsin as reported in the USDA Census of Agriculture between 2012 and 2022. Adoption is presented as a proportion of all harvested cropland in Wisconsin.

For the upper end of CC + NT adoption, we assume that all of the non-harvested cropland has cover crop and no-till adopted. This is unlikely to occur, and assumes that sequestration benefits from the two practices are additive when combined which may not be the case. However, this scenario is used to illustrate the maximum soil carbon sequestration potential of CC + NT in the state.

Under the lower adoption scenario, cover crops and no-till can offset up to 1% of current agricultural sector emissions; under 100% adoption, these practices can offset up to 6% of emissions (Fig. 11).

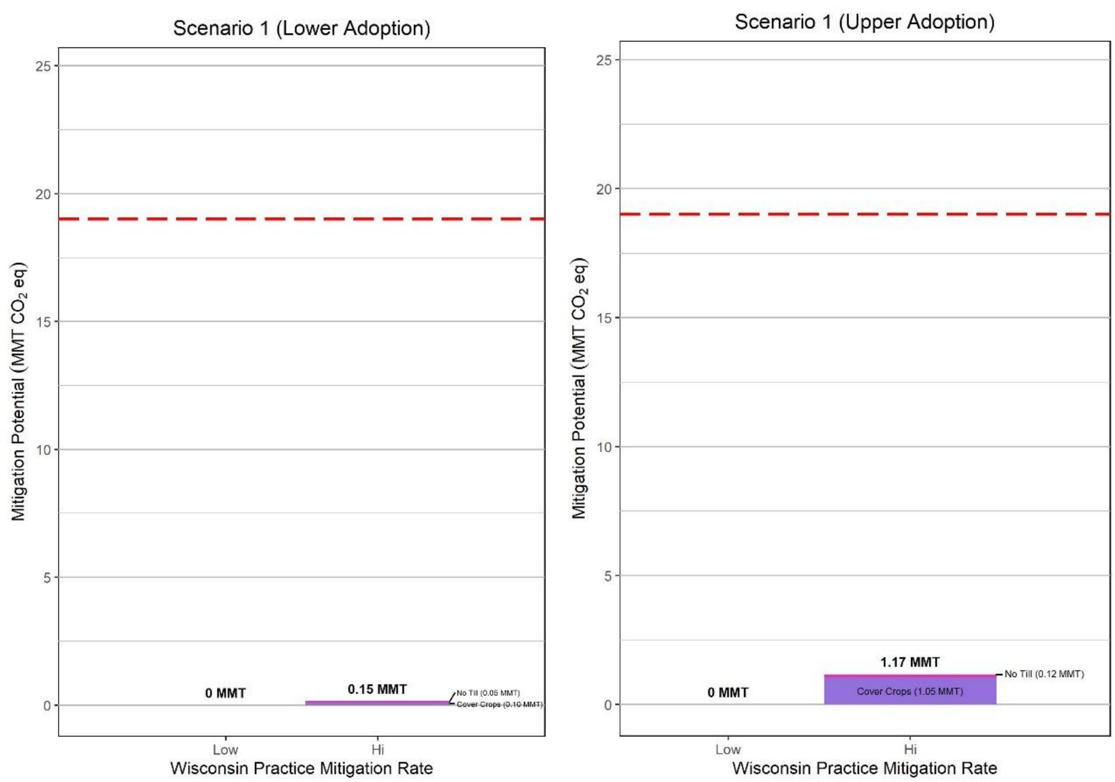

Figure 11. Greenhouse gas mitigation potential under Lower Adoption Rate (left) and Upper Adoption Rate (right) for Scenario 1. In the *Lower Adoption Rate,* estimates assume more conservative increases in practice adoption on Wisconsin farms. The *Upper Adoption Rate* uses an optimal upper estimate that assumes complete or nearly-complete adoption across all applicable acreage in Wisconsin. The horizontal dashed red line indicates the total agricultural sector emissions in the 2021 WDNR GHG Inventory. Each scenario includes an upper (*hi*) and lower (*low*) range of mitigation potential estimates for Wisconsin for each agricultural practice in Wisconsin, as described in the methods, above.

As explained in the methods, we are using lower per-acre soil carbon sequestration rate than other studies and analyses. However, even if we assume higher soil carbon sequestration rates, CC + NT alone is insufficient to offset agricultural sector greenhouse gas emissions. For example, we ran a scenario that assumed no-till can sequester 0.40 Mg CO_2eq $ac^{-1}yr^{-1}$ (from Ogle et al. 2019 values for loamy soils in cool, humid climates and similar to COMET planner estimates) and cover crops sequester 0.49 Mg CO_2eq $ha^{-1}yr^{-1}$ (from the commonly-used Poeplau & Don 2015). In this scenario, cover crops and no-till applied to all available harvested cropland not currently using these practices will sequester enough carbon to offset 23% of agricultural sector emissions.

3.2. Scenario 2: Cover Crop + No Till + Nitrogen Management

For the lower end of nitrogen management, we assume that nitrogen fertilizer use in Wisconsin stabilizes at 2016 levels, the level used by the WDNR inventory we use as a baseline in this analysis. Nitrogen use fluctuates year to year, but has shown a generally increasing trend since 1990 (Fig. 12). Thus, a lower end scenario of not increasing nitrogen fertilizer use is a minimal step towards climate-smart agriculture in Wisconsin by avoiding increased GHG emissions from its use by 2050.

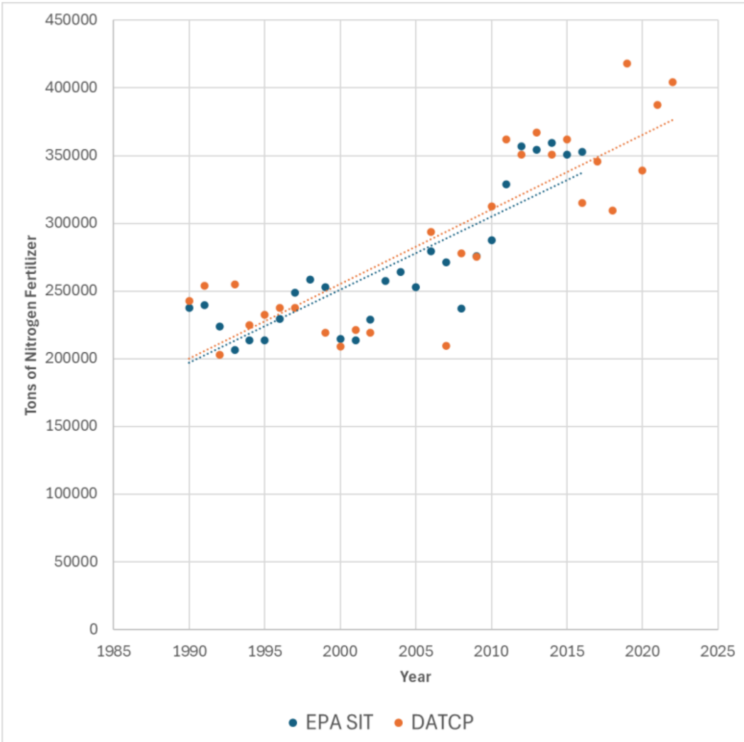

Figure 12. Trends in nitrogen fertilizer use in Wisconsin from two data sources. The EPA state inventory (SIT) tool provides annual estimates of nitrogen fertilizer use in Wisconsin. The DATCP data are annual total nitrogen fertilizer sales in the state. However, not all sales indicate use in the state. Although the DATCP data includes residential use, the vast majority of the use is agricultural.

For the upper end of nitrogen management, we assume a 20% reduction in nitrogen fertilizer use through the four "right" principles for nutrient use: right source, right rate, right time, and right place. Prior analyses in the United States and Canada estimate that these principles can reduce nitrogen fertilizer use by 17 to 22% (Fargione et al. 2018, Drever et al. 2021).

Under the lower adoption rates, CC+NT+Nitrogen Fertilizer management can offset up to 1% of current agricultural sector emissions; under upper adoption rates, these practices can offset up to 9% of emissions (Fig. 13).

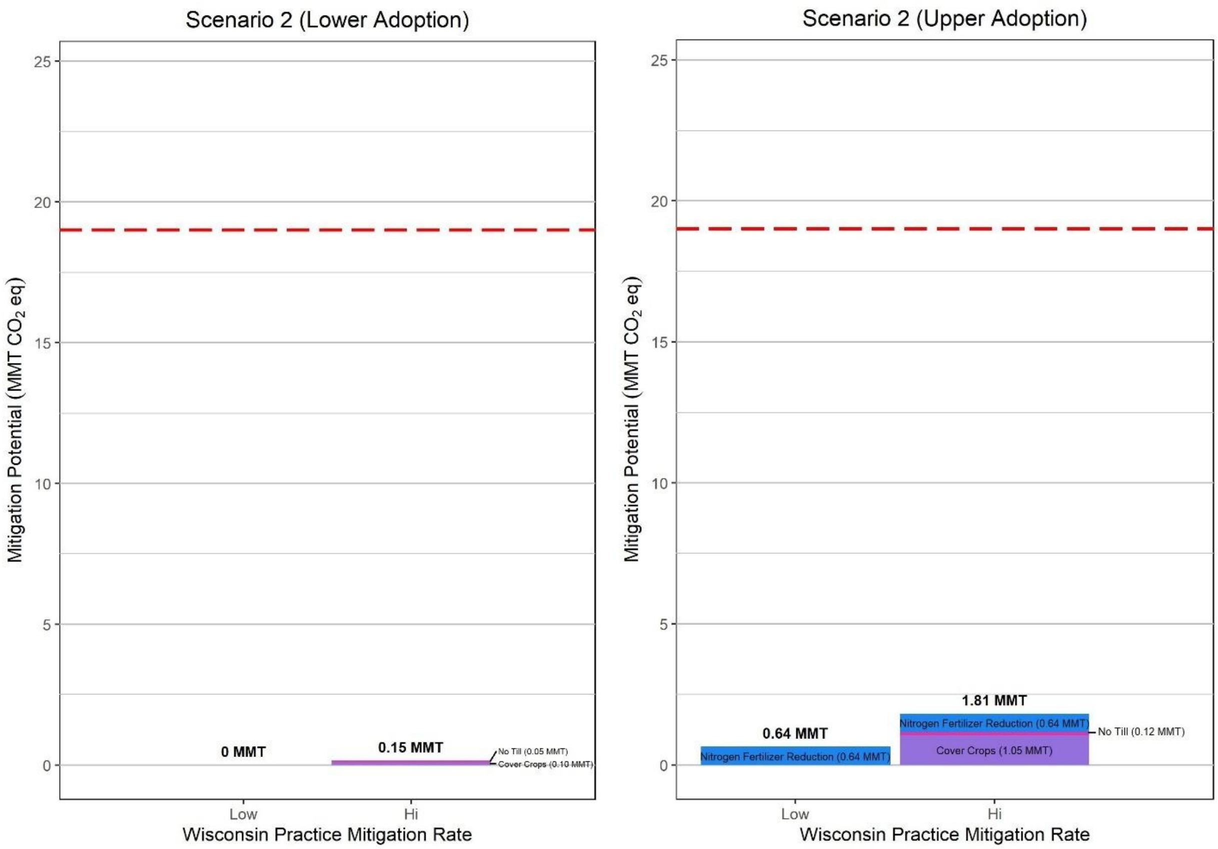

Figure 13. Greenhouse gas mitigation potential under Lower Adoption Rate (left) and Upper Adoption Rate (right) for Scenario 2**.** In the *Lower Adoption Rate,* estimates assume more conservative increases in practice adoption on Wisconsin farms. The *Upper Adoption Rate* uses an optimal upper estimate that assumes complete or nearly-complete adoption across all applicable acreage in Wisconsin. The horizontal dashed red line indicates the total agricultural sector emissions in the 2021 WDNR GHG Inventory. Each scenario includes an upper (*hi*) and lower (*low*) range of mitigation potential estimates for Wisconsin for each agricultural practice in Wisconsin, as described in the methods, above.

3.3. Scenario 3: Cover crop + no-till + nitrogen management + manure management

We developed three dairy cow manure management scenarios that could reduce greenhouse gas emissions (Table 4). We focus only on dairy cows because dairy emissions account for 96% of the state's total livestock manure storage emissions, and 75% of emissions from livestock manure applied to soils in the state. We note that additional manure management greenhouse gas reductions could be seen by addressing manure in other types of livestock in the state.

First, we assumed that all manure currently managed with anaerobic lagoons is first processed with solid-liquid separation. The amount of volatile solids that are removed :depends on the type of separation system used, but here we assume 40% removal (Aguirre-Villegas et al. 2019). To model this within the EPA SIT framework, we re-assign 40% of the volatile solids currently assigned to anaerobic lagoons to solid storage.

For our second manure scenario, we assumed that all manure currently managed with anaerobic lagoons is covered and flared. To model the GHG emission reductions from covering and flaring, we use data from Wrightman and Woodbury (2017), who investigated the GHG emission reductions of these systems in New York state, which has a relatively similar climate to Wisconsin. They found that covered lagoons had a methane conversion factor (MCF) of 0.61 and that flares had an annual efficiency of 81% (i.e., 19% of methane generated is still released. The CO_2 created by the flaring is considered neutral in these calculations since it represents the same carbon that the cow ate (Wrightman, pers. comm.). Thus, we assume that covered and flared lagoons had an effective methane conversion factor of 0.12 (0.61 MCF of covered lagoon x 19% of methane not destroyed by flaring), a significant improvement on the assumed MCF of 0.67 for uncovered lagoons.

For our third scenario, we assumed that all farms with more than 1,000 milk cows used anaerobic digesters to handle manure, and that the remaining anaerobic lagoons on smaller farms were covered and flared. Anaerobic lagoons are a costly investment, so we only applied them to larger farms. Milk herd size breakpoints in USDA Census of Agriculture estimates are 500, 1,000, and 2,500 head. Applying digesters to farms between 500 and 1,000 head is likely too economically burdensome, while only applying them to farms with more than 2,500 head underestimates the climate mitigation potential of digesters in a scenario designed to be as optimistic as possible.

To model this, we need to appropriately shift manure currently managed on large farms from existing manure management systems into anaerobic digesters. However, the EPA SIT used by the WDNR in its GHG inventorying does not break down manure management by farm size, and thus this required making some assumptions subject to error.

To illustrate, the EPA SIT assumes that 24% of all dairy cow manure in Wisconsin is handled in anaerobic lagoons. Aguirre-Villegas and Larson (2017), the only source reporting manure management by farm size in Wisconsin that we are aware of, report that 80% of dairy farms with over 1000 animal units (more than 700 cows) use liquid manure management, which creates a conflict when attempting to apportion manure in this scenario.

According to the USDA Census of Agriculture, 36% of dairy cows in Wisconsin are on farms with 1,000 or more animals. If 80% of this manure is handled by anaerobic lagoons per the estimate from Aguirre-Villegas and Larson, this would mean that at a minimum 29% of manure in the state would be managed by lagoons. This already exceeds the dairy cow manure assigned to lagoons in the EPA SIT, before even accounting for any use of lagoons by smaller farms.

To overcome this discrepancy, we made the following assumption to work within the framework of the SIT. We assumed that 95% of the dairy cow volatile solids assigned to anaerobic lagoons and 100% of the volatile solids assigned to anaerobic digesters comes from farms with 1,000 head. Further, we assumed that the remainder of volatile solids from farms with over 1,000 head are currently assigned to deep pit management in the SIT (i.e. large farms only use digesters, lagoons, and deep pits for manure management).

Table 4. Summary of manure management scenarios that could reduce greenhouse gas emissions, along with avoided emissions.

Manure Management System	Percent of dairy cow volatile solids managed			
	Current	Manure Management Scenario 1: Solid Liquid Separation	Manure Management Scenario 2: Cover and Flare	Manure management Scenario 3: Anaerobic Digester Adoption and Flaring
Anaerobic Digester	5.9%	5.9%	5.9%	36.2%
Anaerobic Lagoon	23.7%	14.2%	0.0%	0.0%
Daily Spread	5.4%	5.4%	5.4%	5.4%
Deep Pit	22.7%	22.7%	22.7%	14.9%
Liquid/Slurry	3.2%	3.2%	3.2%	3.2%
Pasture	14.9%	14.9%	14.9%	14.9%
Solid Storage	24.2%	33.6%	24.2%	24.2%
Cover and Flare	-	0.0%	23.7%	1.2%
State-Weighted MCF	23.02	16.82	9.89	6.20
Reduced Emissions ($MMTCO_2eq$)		0.75	1.86	2.66
% Manure Management Emissions Reduced		15%	37%	53%
% Ag Sector Emissions Reduced		4%	9%	13%

There are some important limitations of this manure management analysis that can hopefully be addressed in future efforts. These limitations include:

- This analysis would be improved with a better understanding of manure management strategies in the state, broken down by farm size.
- This analysis does not include the effect of management shift on heifer manure, which is calculated separately from milk cow manure in the SIT. If farms manage milk cow and heifer manure concurrently with the same systems, this should be reflected in the emissions inventory. Currently, the EPA SIT assumes that all heifer manure in Wisconsin is managed as a dry lot, to which an MCF of 1% is applied, providing no room for improvement from a climate perspective.
- This analysis does not include the potential benefit of solid liquid separation on the back end of digesters. The MCF for anaerobic digesters includes both leakage and emissions from the storage of digestate. We are not aware of any MCF estimates that include SLS of the digestate. However, the benefit of this practice has been quantified in a different modeling framework (Aguirre-Villegas et al. 2019).

- This analysis does not include any changes in N_2O emissions due to the digestion process in anaerobic digesters. This is not currently included in IPCC recommended methods.
- This analysis does not include the climate benefits of electricity generated from anaerobic digesters displacing fossil fuel-generated electricity. On-farm electricity use is considered in a different module of the EPA SIT.
- The EPA SIT used by the WDNR in its GHG inventorying uses the 100-year methane global warming potential (GWP) of 25 rather than the 20-year GWP of 84. To maintain consistency with the WDNR inventory, which we are using for our baseline emissions, we also use the 100-year GWP. However, using the 20-year GWP would both increase the baseline agricultural sector emissions but also magnify the climate benefit of improved manure management.

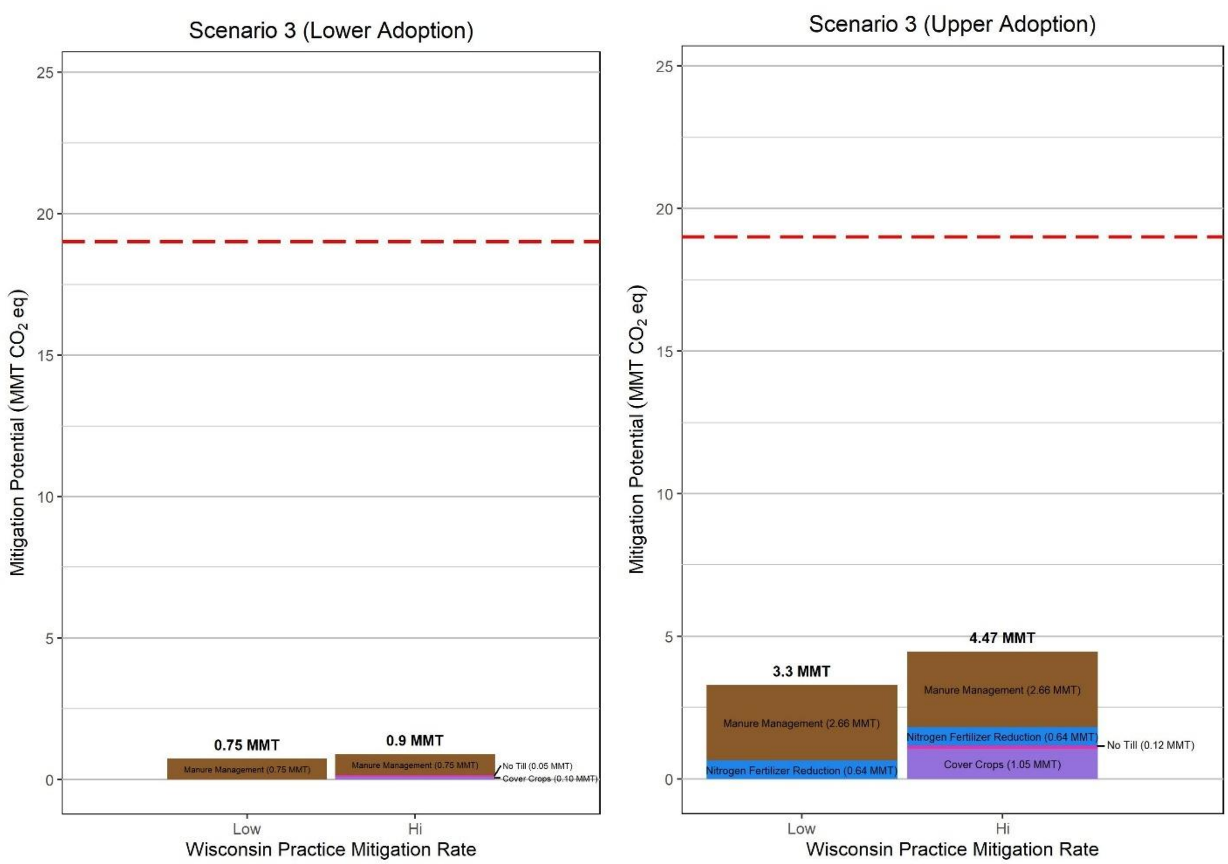

Figure 14. Greenhouse gas mitigation potential under Lower Adoption Rate (left) and Upper Adoption Rate (right) for Scenario 3. In the *Lower Adoption Rate,* estimates assume more conservative increases in practice adoption on Wisconsin farms. The *Upper Adoption Rate* uses an optimal upper estimate that assumes complete or nearly-complete adoption across all applicable acreage in Wisconsin. The horizontal dashed red line indicates the total agricultural

sector emissions in the 2021 WDNR GHG Inventory. Each scenario includes an upper (*hi*) and lower (*low*) range of mitigation potential estimates for Wisconsin for each agricultural practice in Wisconsin, as described in the methods, above.

3.4. *Scenario 4: Cover crop + no-till + nitrogen management + biochar + manure management + biochar amendment + grazing management*

This scenario represents the greatest possible GHG offsets from adoption of practices within the current dominant paradigm of annual row crops and confined milk production.

For the lower and upper potentials of the biochar amendment scenarios, we use the lower and upper estimate of logging residue woody biomass available as biochar feedstock, respectively (see methods). We assume that 60% of current pasture in Wisconsin could benefit from improved grazing management (grazing intensity and grazing frequency). For our lower adoption scenario, we assume that 50% of this pasture improves its grazing management. For our upper adoption scenario, we assume that 100% of this pastureland improves its grazing management.

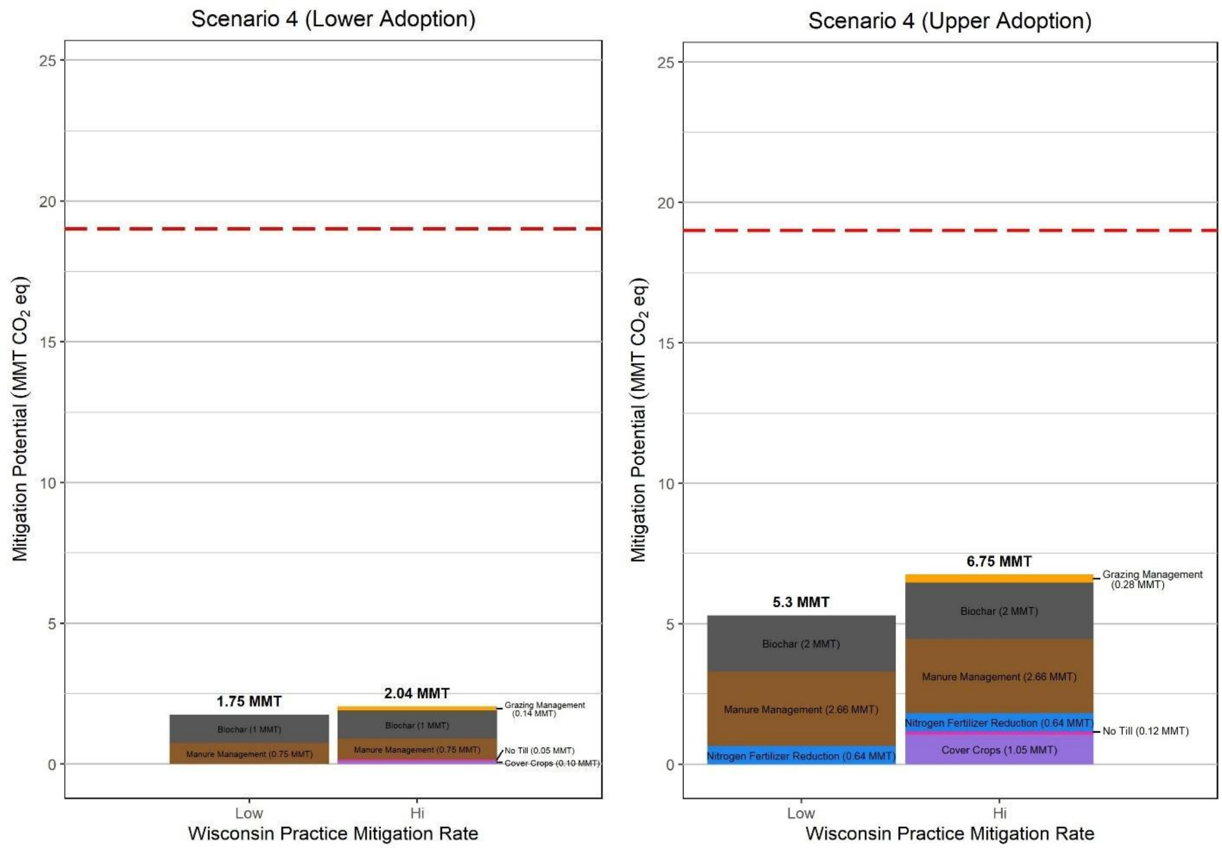

Figure 15. Greenhouse gas mitigation potential under Lower Adoption Rate (left) and Upper Adoption Rate (right) for Scenario 4**.** In the *Lower Adoption Rate,* estimates assume more conservative increases in practice adoption on Wisconsin farms. The *Upper Adoption Rate* uses an optimal upper estimate that assumes complete or nearly-complete adoption across all applicable acreage in Wisconsin. The horizontal dashed red line indicates the total agricultural sector emissions in the 2021 WDNR GHG Inventory. Each scenario includes an upper (*hi*) and lower (*low*) range of mitigation potential estimates for Wisconsin for each agricultural practice in Wisconsin, as described in the methods, above.

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4. SCENARIOS INCORPORATING TRANSITION TO PERENNIAL SYSTEMS

In the following scenarios we model the conversion of annual row crops to herbaceous perennial crops, agroforestry, pastures and solar farms, as well as the introduction of trees into existing pasture for silvopasturing.

4.1 Potential Scope of Row Crop Conversion

The scope of the implementation in this round of scenarios is to avoid impacting any annual crop land needed for livestock feeding in the state. Of all corn grain grown in

Wisconsin, 37% goes to ethanol production, amounting to 1.12 million acres. Another 25% of corn grain is either surplus or exported (Jackson 2024; estimates based on Iowa data). This provides another 743,000 acres of corn not used to feed livestock in the state. Furthermore, 65% of soybeans in Wisconsin are exported (CoolBean 2024), amounting to 1.4 million acres.

All this land provides 3.2 million acres of current soy or corn production that is not used for food or feeding livestock in the state.

The one exception is that when modeling conversion to pasture needed to support a transition from confined to grassfed dairy production, we do take into account the cropland currently used to feed confined cows (see methods).

We also apply an ecological bounding condition where agroforestry is not implemented on land that was prairie in original land-survey records from the mid-1800s. A digitization of state land cover in the mid-1800s from these records was obtained from the Wisconsin Department of Natural Resources (WDNR 2025). Of all the current cropland, approximately 20% had historically been prairie with no trees. Thus, total cropland-to-agroforestry conversion could not exceed 7 million acres. Of all the current pasture, 14% had historically been prairie with no trees. Thus, total pasture-to-silvopasture conversion cannot exceed 963,000 acres

4.2. Scenario 5: Annual Agricultural soils + Perennial system conversion

4.2.1 Conversion of annual crops to grassland in the form of solar farms

Utility-scale solar farms (defined here as greater than 100 MW capacity) in Wisconsin are being developed on former agricultural land and are maintained with perennial vegetation grassland (e.g., side-oats grama, upland bent, little bluestem) underneath and around the panels. Wisconsin needs approximately 200,000 acres of utility scale solar farms to reach carbon-free electricity production targets. For a conservative adoption scenario we assume that half of this target is achieved; for a more aggressive adoption scenario, we assume that the full target is achieved.

4.2.2. Riparian forest buffer establishment

NRCS Standard 391 suggests maximizing widths and lengths of buffers to maximize environmental benefits. It lists a minimum width of 35 feet for sediment and organic matter control, carbon storage and wildlife habitat. It recommends expanding width to 50 feet to reduce nutrient, pesticide, and pathogen runoff and to improve edge habitat. Finally, it recommends 100-foot width for interior forest bird habitat and 165 feet for large mammals.

Using the Wiscland data set (30-meter resolution) we calculated the amount of non-forage agricultural land within 30 m (98 feet) and 60 m (197 feet) of perennial water bodies and primary streams. This amounts to 142,645 and 261,350 acres, respectively for potential conversion to riparian buffer.

For the lower end of our full NCS scenario we assume that half of this area within 30 m is converted, to approximate 50-foot forest buffer widths. For the upper end we assume conversion of all current non-forage agricultural land within 60m to riparian forest buffer.

4.2.3 Windbreak Establishment

For the lower end we follow the approach of Fargione et al. (2018) to calculate the windbreak opportunity area as being 5% of wind-erosion prone acres in the state, which amounts to 77,000 acres (0.88% of current cropland). For the upper end, we assume that benefits of windbreaks go beyond erosion control to include increased crop production and homestead sheltering. Following Ballesteros-Possu et al. (2017), we assume windbreaks are established on 5% of cropland, with 5% being identified as the threshold for economic advantage of windbreaks.

4.2.4 Silvopasture Adoption

For a lower end, we assume that 10% of existing pasture can be converted to silvopasture, following Udawatta & Jose (2010), as done by Fargionne et al. (2018). For the upper end we assume introduction of trees onto 50% of all current pasture land occurring in historically forested or savanna land.

4.2.5 Conversion of annual crops to herbaceous perennial crops (e.g., Kernza®, alfalfa, switchgrass)

For a lower estimate, we assume that conversion to perennial herbaceous crops will reach the current acreage of an established non-corn or soy crop. The two most prominent non-corn or soy field crops are wheat (240,000 acres) and oats (65,000 acres). We thus assume a lower end of conversion to perennial herbaceous crops to 240,000 acres.

Assuming the upper end conversion for solar farms, riparian buffers and windbreaks, as well as the lower end conversion to perennial herbaceous crops and alley crops, there are still 1.2 million acres of non-feed corn and soy that could be converted. For the upper end of perennial herbaceous crops and alley crops, we apportion the remaining 1.2 million acres equally between the two practices.

4.2.6 Alley crop establishment

For a lower end, we assume that 10% of cropland is converted to alley crops, following Fargione et al. (2018) and Drever et al. (2021). Assuming the upper end conversion for solar farms, riparian buffers and windbreaks, as well as the lower end conversion to perennial herbaceous crops and alley crops, there are still 1.2 million acres of non-feed corn and soy that could be converted. For the upper end of perennial herbaceous crops and alley crops, we apportion the remaining 1.2 million acres equally between the two practices.

This conversion contemplates establishing strips of tree crops (e.g., fruit or nut trees, trees desired for wood) within larger crop fields of annual or perennial herbaceous crops.

Table 5. Summary of total acres and rationale for NCS practice adoption used in our analyses under the low and high adoption scenarios. Conversion for most practices here refers to conversion of current corn and soybean acreage not currently used for livestock or human feed (3.2 million total acres) to each NCS practice listed. The exceptions are silvopasture, which represent the acres of existing pasture that trees are added to, and grazing optimization, which refers to the number of current pasture acreage (1.1 million total acreage) that could have improved grazing management. See Table A.19 for more specific inputs into each scenario

NCS Practice	Lower Adoption Rate (acres)	Brief Rationale	Upper Adoption Rate (acres)	Brief Rationale
Conversion of annual cropland to perennial row crops	240,000	Equivalent to an established commodity crop (wheat)	840,000* *240,000 when including 47% transition to grassfed dairy	Replacing rest of available corn and soybean acres not used for livestock feed in the state
Conversion of annual row crops to solar arrays maintained with native grasses	100,000	Acreage needed for 50% implementation of utility scale solar needed for 100% carbon free electricity generation in state	200,000	Acreage needed for full implementation of utility scale solar needed for 100% carbon free electricity generation in state
Forested riparian buffer establishment	71,323	Non-forage agricultural land within 50 feet of waterbodies	261,350	Non-forage agricultural land within 200 feet of waterbodies
Windbreak establishment	77,000	5% of erosion-prone cropland in the state	438,000	5% of all cropland using economically-beneficial threshold
Alley cropping	876,000	10% of current cropland	1,476,000 * *876,000 when including 47% transition to grassfed dairy	Replacing rest of available corn and soybean acres not used for livestock feed in the state
Silvopasture	112,000	10% of existing pasture	564,000	60% of existing pasture on historically forested or savanna land
Grazing management	335,764	30% of existing pasture	671,527	60% of existing pasture
Expanded pasture from transitioning dairy production to grassfed	644,444	Transitioning 25% of current milk production	1,200,000	Transitioning 47% of current milk production
"Conservation" agriculture practices	Cover crops: 573,472 No-till: 1,907,040	Projection from 2012-2022 trends	Cover crop: 1.8m – 2.7m No-till: 160k – 1m	100% adoption of cover crop and no-till practices on all harvested annual cropland remaining, following conversion to NCS crops in a given scenario
Nitrogen management	Nitrogen fertilizer application reduction from converting annual row crop acreages as outlined in each scenario to NCS crops + a 20% reduction in nitrogen use on remaining cropland			
Biochar	Annual application of 420,000-840,000 tons of biochar to remaining cropland (at 0.2 tons per acre)			

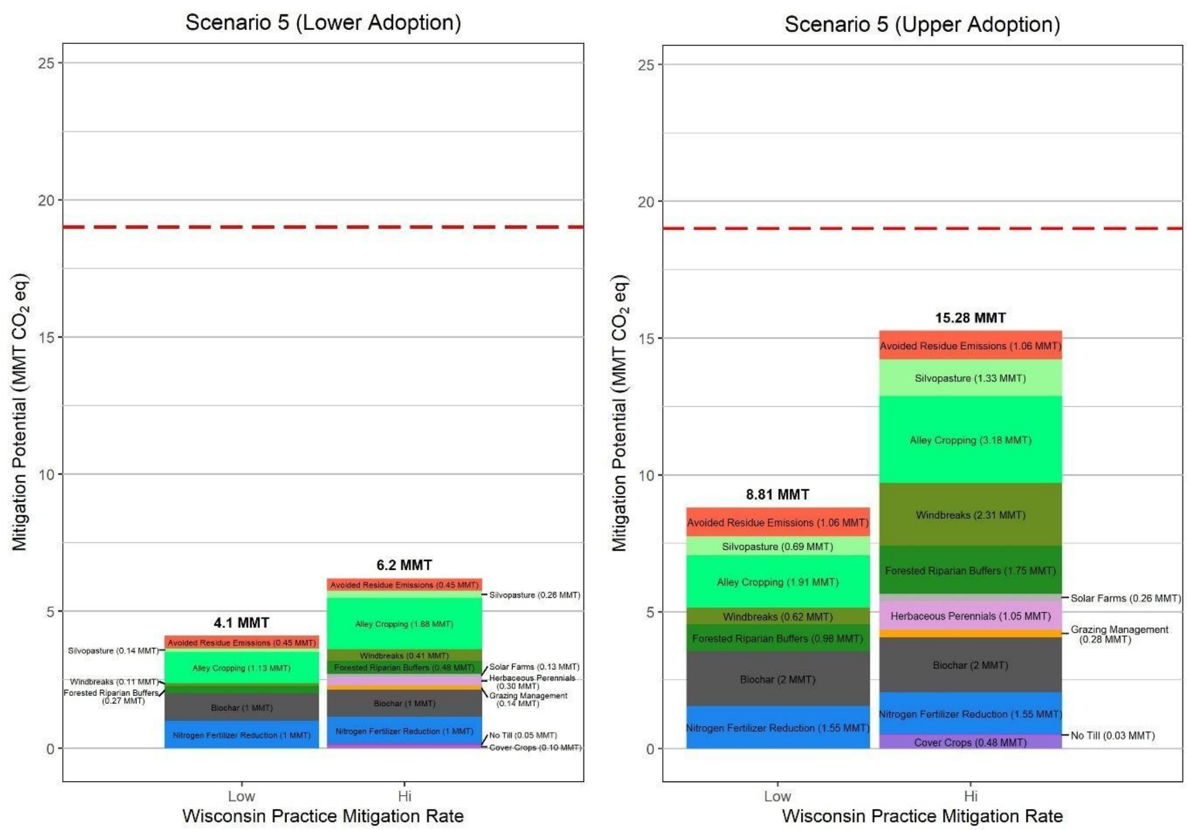

Figure 16. Greenhouse gas mitigation potential under Lower Adoption Rate (left) and Upper Adoption Rate (right) for Scenario 5. In the *Lower Adoption Rate,* estimates assume more conservative increases in practice adoption on Wisconsin farms. The *Upper Adoption Rate* uses an optimal upper estimate that assumes complete or nearly-complete adoption across all applicable acreage in Wisconsin. The horizontal dashed red line indicates the total agricultural sector emissions in the 2021 WDNR GHG Inventory. Each scenario includes an upper (*hi*) and lower (*low*) range of mitigation potential estimates for Wisconsin for each agricultural practice in Wisconsin, as described in the methods, above.

4.2.7 Scenario 6: Annual Agricultural soils + Perennial system conversion + Manure Management

In Scenario 6 we add manure management to Scenario 5, using increased use of solid liquid separation for the lower adoption scenario and replacing anaerobic lagoons with anaerobic digesters on large farms, while covering and flaring the remaining anaerobic lagoons as the upper adoption scenario.

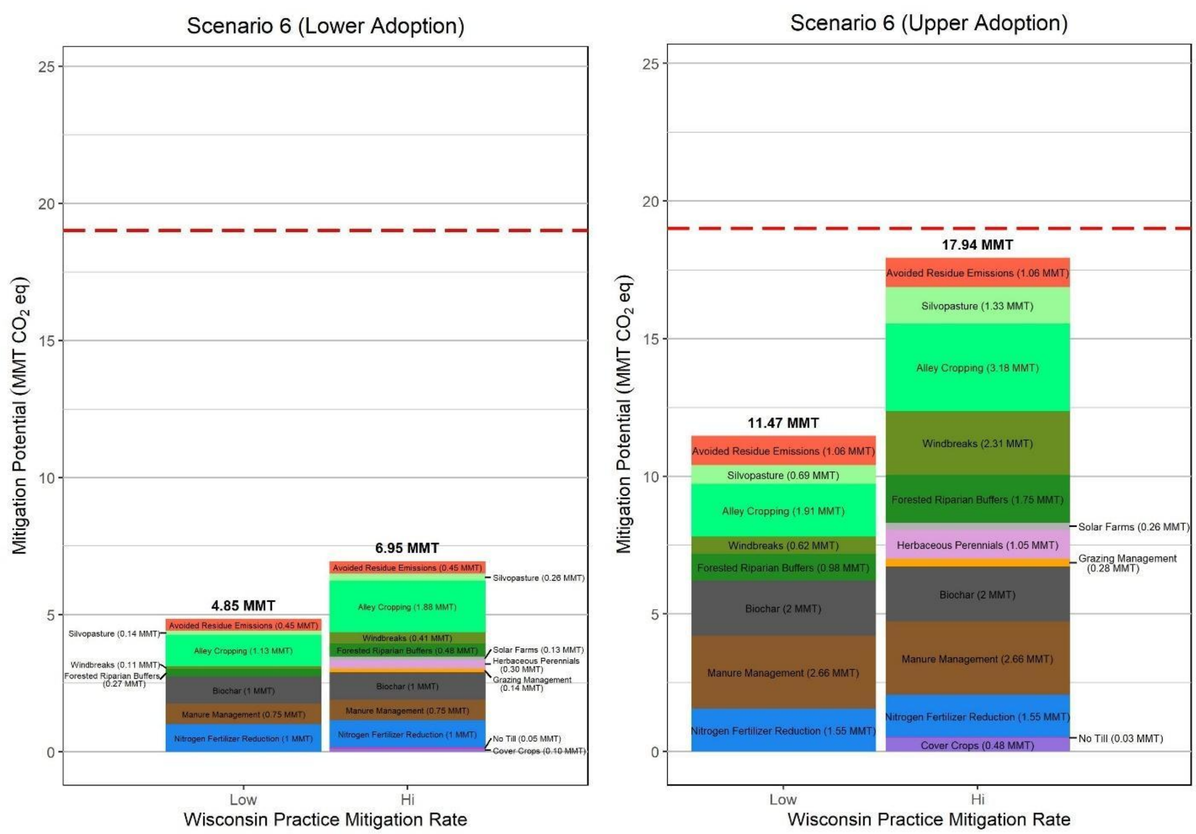

Figure 17. Greenhouse gas mitigation potential under Lower Adoption Rate (left) and Upper Adoption Rate (right) for Scenario 6. In the *Lower Adoption Rate,* estimates assume more conservative increases in practice adoption on Wisconsin farms. The *Upper Adoption Rate* uses an optimal upper estimate that assumes complete or nearly-complete adoption across all applicable acreage in Wisconsin. The horizontal dashed red line indicates the total agricultural sector emissions in the 2021 WDNR GHG Inventory. Each scenario includes an upper (*hi*) and lower (*low*) range of mitigation potential estimates for Wisconsin for each agricultural practice in Wisconsin, as described in the methods, above.

5. *SCENARIOS INCLUDING TRANSITION TO GRASSFED MILK PRODUCTION*

Our first grassfed transition scenario assumed that milk production in the state remained constant, but 25% (lower adoption) to 47% (upper adoption[9]) of the milk production was shifted to 100% grassfed. There is a net increase in cradle-to-farm gate GHG emissions for the milk production when shifting from shifting 25-47% of milk production from confined to

[9] 47% was chosen rather than 50% as it is the amount of production that can be shifted to grassfed on the 1.6 million acres of non-livestock feed corn and soy available after the lower end of our NCS practice adoption.

grassfed (Table 6). However, this is more than offset by potential soil carbon sequestration in land converted to pasture and by avoided nitrogen fertilizer emissions and crop residue emissions on land not currently used for dairy production that is converted to pasture. Assuming no soil carbon storage there is a slight decrease (0.07-0.13 MMT CO_2eq) in GHG emissions associated with this shift; assuming our upper end of soil carbon sequestration, shifting 25% of milk production to grassfed will reduce agricultural sector GHG emissions by 1.18 MMT CO_2eq and shifting 47% of milk production to grassfed will reduce agricultural sector GHG emissions by 2.22 MMT CO_2eq.

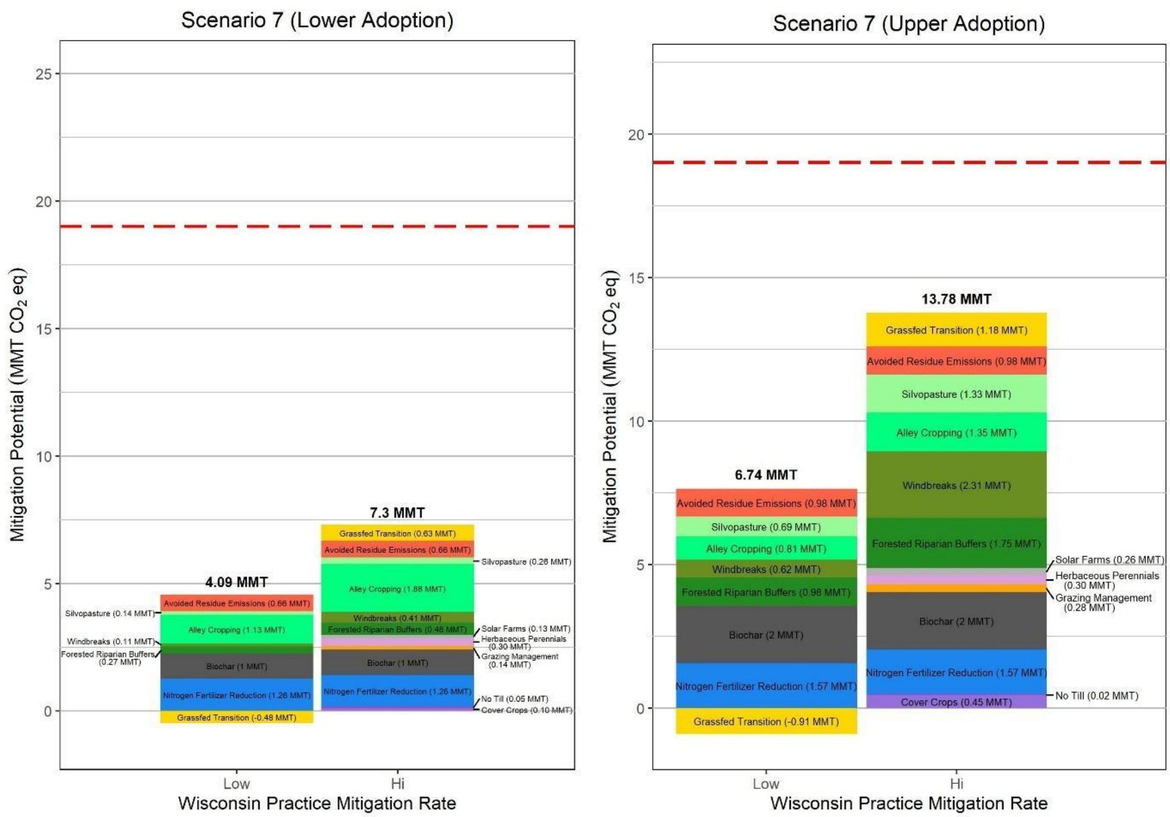

Figure 18. Greenhouse gas mitigation potential under Lower Adoption Rate (left) and Upper Adoption Rate (right) for Scenario 7. In the *Lower Adoption Rate,* estimates assume more conservative increases in practice adoption on Wisconsin farms. The *Upper Adoption Rate* uses an optimal upper estimate that assumes complete or nearly-complete adoption across all applicable acreage in Wisconsin. The horizontal dashed red line indicates the total agricultural sector emissions in the 2021 WDNR GHG Inventory. Each scenario includes an upper (*hi*) and lower (*low*) range of mitigation potential estimates for Wisconsin for each agricultural practice in Wisconsin, as described in the methods, above.

Our second grassfed transition scenario assumed that milk production in the state was limited to milk produced by grassfed cows while maintaining the current milk-cow herd size. The current herd size of 1.27 million milk cows would produce ~8.43 billion kg of milk (a 42% reduction from current production levels). This results in a 5.17 MMT CO_2eq reduction in GHG emissions. This also leads to the conversion of 1.5 million acres of corn and soybean fields to pasture, providing 0 to 2.0 MMT CO_2eq of soil carbon sequestration, as well as 0.59 MMT of avoided GHG emissions from nitrogen fertilizer applications and crop residues on corn/soy land converted to pasture not currently used for dairy production.

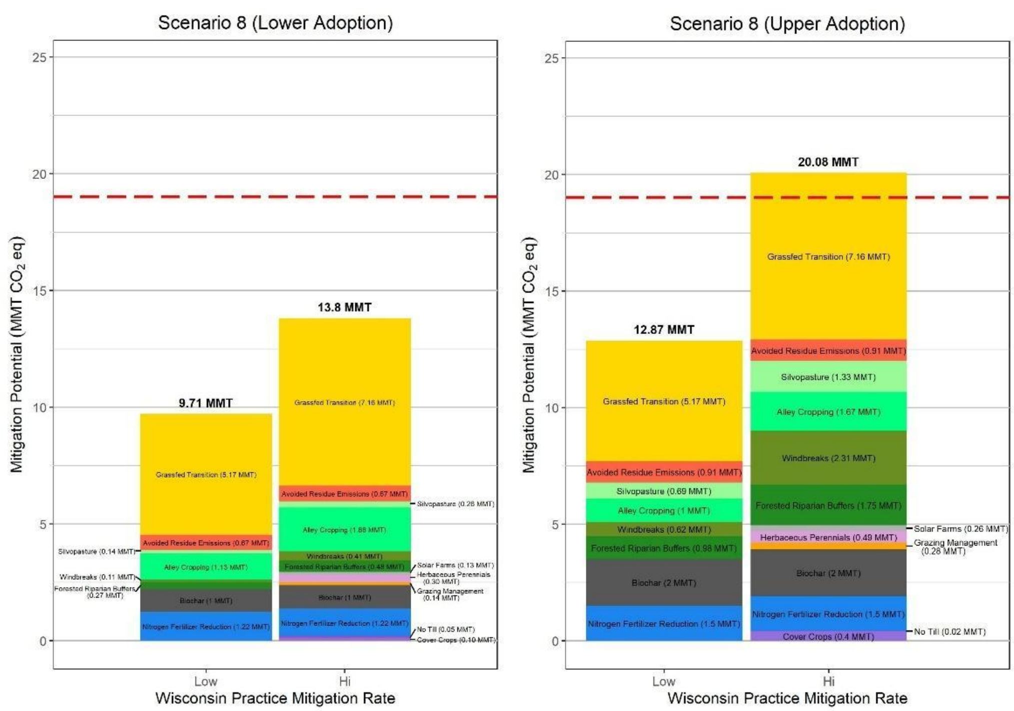

Figure 19. Greenhouse gas mitigation potential under Lower Adoption Rate (left) and Upper Adoption Rate (right) for Scenario 8. In the *Lower Adoption Rate,* estimates assume more conservative increases in practice adoption on Wisconsin farms. The *Upper Adoption Rate* uses an optimal upper estimate that assumes complete or nearly-complete adoption across all applicable acreage in Wisconsin. The horizontal dashed red line indicates the total agricultural sector emissions in the 2021 WDNR GHG Inventory. Each scenario includes an upper (*hi*) and lower (*low*) range of mitigation potential estimates for Wisconsin for each agricultural practice in Wisconsin, as described in the methods, above.

Our final grassfed transition scenario assumed all milk production was limited to milk produced by grassfed cows on the land area currently used for dairy production. The land currently used for dairy production can support 940,000 grassfed milk cows, producing 6.25 billion kg of milk (a 57% reduction from current production levels). This results in a 7.75 MMT CO_2eq reduction in GHG emissions. This also leads to the conversion of 854,000 acres of corn and soybean fields to pasture, providing 0 to 1.0 MMT CO_2eq of soil carbon sequestration.

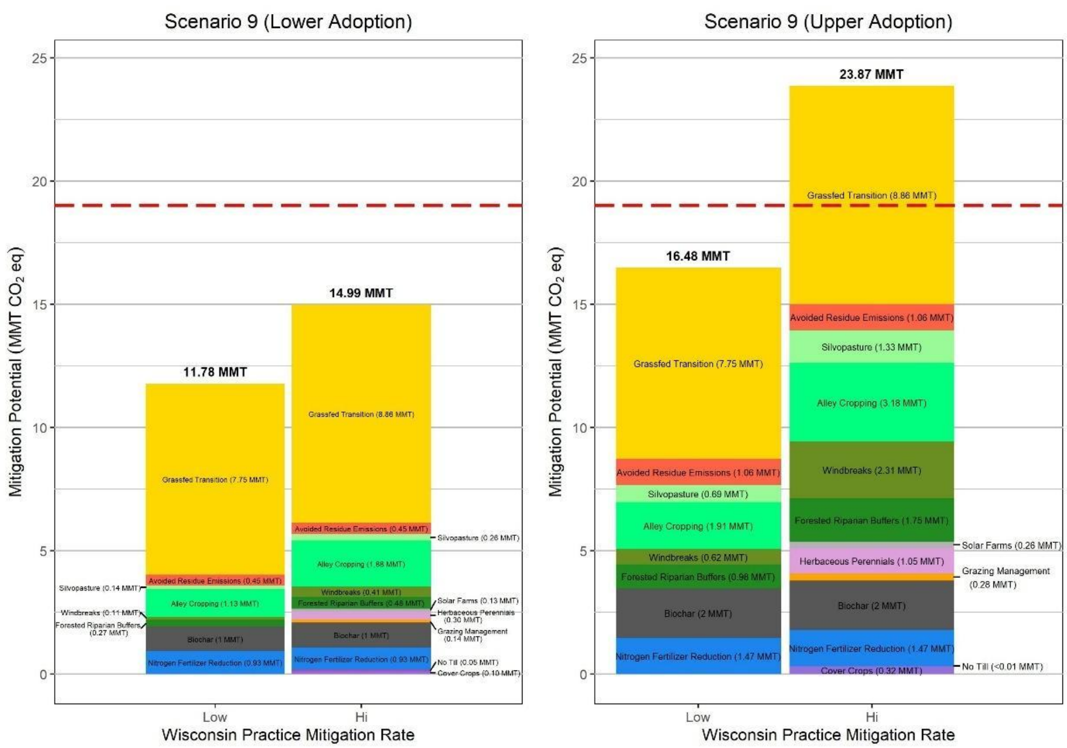

Figure 20. Greenhouse gas mitigation potential under Lower Adoption Rate (left) and Upper Adoption Rate (right) for Scenario 9. In the *Lower Adoption Rate,* estimates assume more conservative increases in practice adoption on Wisconsin farms. The *Upper Adoption Rate* uses an optimal upper estimate that assumes complete or nearly-complete adoption across all applicable acreage in Wisconsin. The horizontal dashed red line indicates the total agricultural sector emissions in the 2021 WDNR GHG Inventory. Each scenario includes an upper (*hi*) and lower (*low*) range of mitigation potential estimates for Wisconsin for each agricultural practice in Wisconsin, as described in the methods, above.

When incorporating the shift of 47% of milk production to grassfed (Scenario 7) or moving the current milk cow herd to grassfed (Scenario 8) into the existing perennial transition scenario (scenario 5), we reduced the acres of adoption of alley cropping and transition to perennial herbaceous crops to accommodate the land needed for pasture conversion. Conversion to alley crops and perennial herbaceous had the lowest GHG mitigation potential of the conversions we included in this analysis.

Finally, for all scenarios that include a transition to grassfed dairy production, we do not also include improved manure management in the remaining confinement dairies. Thus, additional greenhouse gas reductions could be achieved in these scenarios by improving manure management in the remaining confinement dairies.

Table 6. Changes in milk production, land use, and greenhouse gas emissions under alternative transition-to-grassfed dairy scenarios.

	Shift 25% of milk production to grassfed	**Shift 47% of milk production to grassfed**	**Maintain current herd size**	**Maintain current landbase**
Milk production (Mg/yr)	14,438,230	14,438,230	8,434,070	6,250,061
Change in milk cows (head)	+226,035	+343,668	0	-328,832
Change in milk production (%)	0	0	-42%	-57%
Total corn/soy converted to pasture (acres)	857,873	1,612,799	1,541,317	853,768
Existing dairy corn/soy converted to pasture (acres)	213,443	401,271	853,768	853,768
Other non-feed corn/soy converted to pasture (acres)	644,430	1,211,528	687,549	0
Reduction in cradle-to-farm gate milk production emissions allocated to the agricultural module (MMT CO_2eq per year)[1]	-0.48	-0.91	5.17	7.75
Soil carbon sequestration potential from corn/soy converted to pasture (MMT CO_2eq/yr)	0-1.11	0-2.09	0-2.0	0-1.11
Estimated avoided N fertilizer GHG emissions from non-feed corn/soy converted to pasture (MMT CO_2eq/yr)2	0.34	0.64	0.36	0
Estimated avoided corn/soy residue emissions from non-feed corn/soy converted to pasture (MMT CO_2eq/yr)2	0.21	0.40	0.23	0
Total Ag Sector GHG Offset (MMT CO_2eq/yr)	**0.07-1.18**	**0.13-2.22**	**5.76-7.76**	**7.75-8.86**

[1] Assuming 1.28 kg CO_2eq per kg FPCM produced in confinement and 1.46 kg CO_2eq per kg FPCM produced by 100% grassfed cows (see methods)

[2] Avoided emissions from fields currently used for dairy production already included in the cradle-to-farm gate carbon intensity calculations.

6. *CLOSING THE GAP WHILE MINIMIZING MILK PRODUCTION REDUCTIONS*

In the above scenarios, the only ones that fully offset current agricultural sector emissions are those that involve significant reductions in milk production (Scenarios 8 and 9). Our most optimistic scenario without milk production reductions offsets 94% of greenhouse gas emissions in the agricultural sector (Scenario 6; Fig 9b). To close this gap, we consider two pathways. First, we look at reducing enteric emissions from dairy production through dietary supplements. As noted in the methods, the long-term effectiveness of dietary supplements in reducing enteric emissions is understudied, leading to questions about the longevity of the observed reductions (see methods for more details). However, recent meta-analyses of short-term trials of supplementing diets with 3-NOP report enteric emissions reductions of over 30% (e.g., Kebreab et al. 2023). Using this level of reduction is recommended in the recent USDA "blue book" of GHG accounting (Hanson et al. 2024).

To achieve a 100% offset in GHG emissions from the agricultural sector, it would take a 24% reduction in enteric emissions added to Scenario 6. This is not out of the realm of possibility, although longer-term studies would be needed to evaluate the long-term efficacy of 3-NOP diet supplement on enteric emissions reductions. Another area showing promise for permanent reductions is breeding for lower methane emissions. There is evidence that enteric reductions up to 24% from selective breeding are possible by 2050 (Bell et al. 2010; de Haas et al. 2021) which would also be enough to close the gap to net-zero from Scenario 6.

Second, we look at what level of milk production reduction would be needed in addition to Scenario 6 to reach 100% GHG emissions offset in the agricultural sector. Reduced manure and enteric emissions alone from a 10% reduction in milk production added to Scenario 6 would reach 100% GHG emissions offset.

An estimated 22% of dairy products in the United States are thrown away (Campbell and Feldpausch 2022). Working to reduce this food waste by just half would provide the production reduction needed to reach net-zero GHG emissions in the agricultural sector from reduced manure and enteric emissions alone.

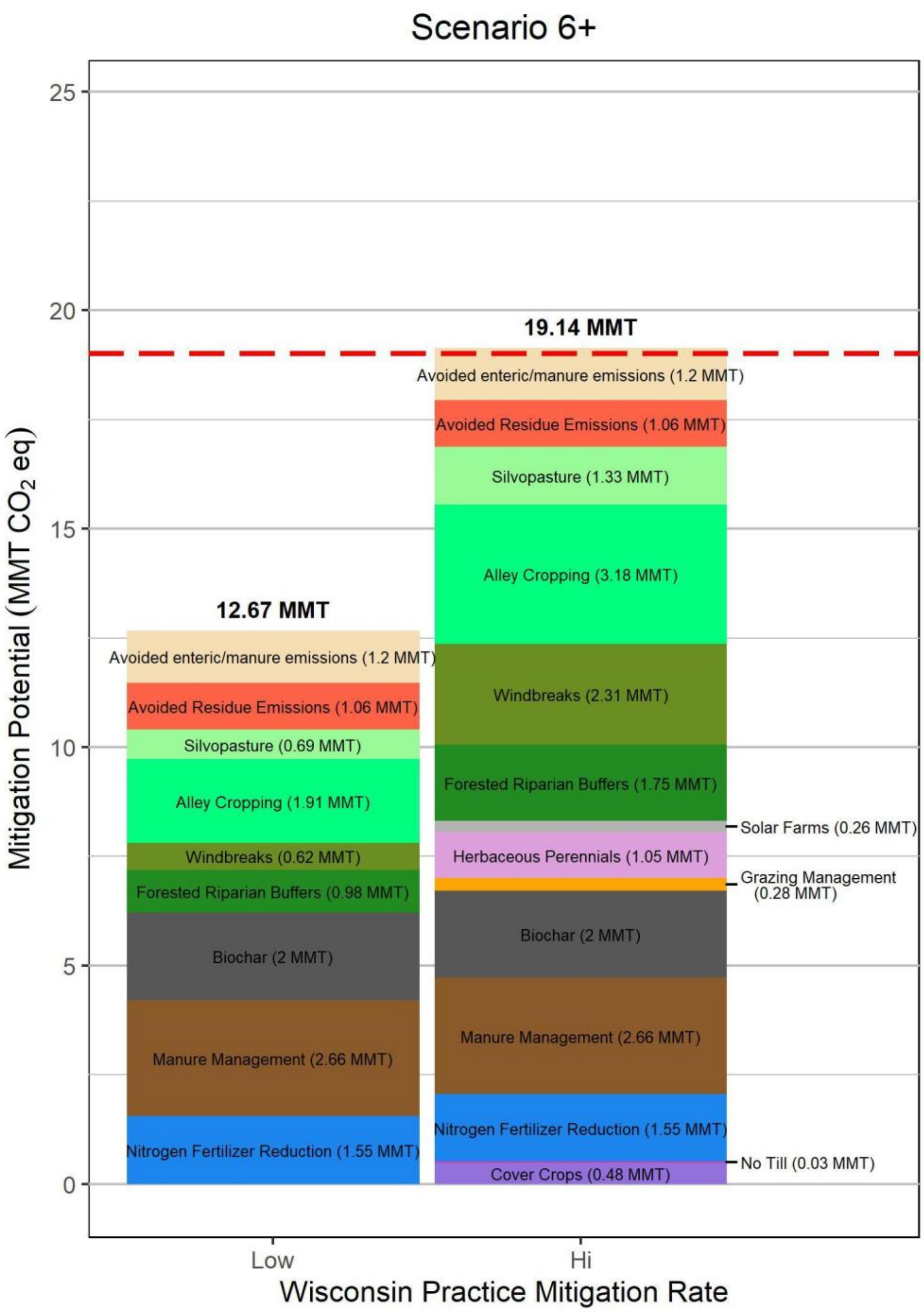

Figure 21. Greenhouse gas mitigation potential under Scenario 6+, where we add to Scenario 6 the avoided manure and enteric emissions reductions from a 10% reduction in milk production in Wisconsin. The horizontal dashed red line indicates the total agricultural sector emissions in the 2021 WDNR GHG Inventory. Each scenario includes an upper (*hi*) and lower (*low*) range of mitigation potential estimates for Wisconsin for each agricultural practice in Wisconsin, as described in the methods, above.

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Table A.1 Summary of soil carbon sequestration potential of adopting no-till as reported in published reviews and meta-analyses. If climate was found to be a significant modifying factor in sequestration potential, appropriate values for WI are provided. Uncertainty presented in terms of standard error.

Study	**SOC Potential (Mg C ac^{-1}yr^{-1})**	**CO_2Eq Potential (Mg CO_2eq ac^{-1}yr^{-1})**	**Scope**	**Depth Measured**
Virto et al. 2012	0.09 (0.02)	0.34 (0.09)	Global, Euro and NA Focus	30cm
Liang et al. 2019	Insignificant[1]	--	Eastern Canada, Wet and Cool Climate	60cm
Meurer et al. 2018	Insignificant	--	Global, Boreo-temperate regions	150cm
Ogle et al 2019	0.11 (0.06)	0.40 (0.22)	Global Cool, Moist, and Loamy, Silty, and Clayey soils[2]	30cm
Haddayway et al. 2017	Insignificant	--	Global, Boreo-temperate regions	150cm
Luo et al. 2010	Insignificant	--	Global	60cm, with data down to 140cm
Drever et al. 2021	0.01-0.06	0.03-0.21	Canadian provinces	Not reported
COMET-Planner	0.11(Red Till[a])/ 0.14(Int Till[b])	0.41(Red Till[3]) / 0.52(Int Till[4])	Averaged across WI counties	30cm

[1] - Study covers Eastern and Western Canada; reported Eastern Canada results due to proximity of Eastern Canada to Wisconsin

[2] - Note that the confidence intervals for this estimate include 0.

[3] - Reduced Till: increased carbon sequestered by switching from reduced tillage to no tillage.

[4] - Intensive Till: increased carbon sequestered by switching from intensive tillage to no tillage.

Table A.2. Summary of potential soil organic carbon (SOC) gains from implementation of cover crops as reported in published reviews and meta-analyses of field-based experimental measurements. If climate was found to be a significant modifying factor in sequestration potential, appropriate values for WI are provided in footnote. Uncertainty presented in terms of standard error. We have also included estimates for Wisconsin based on COMET-Planner's process-based modeling of carbon cycling informed by land management, soil properties, and climate conditions.

Study	**SOC Potential (Mg C $ac^{-1}yr^{-1}$)**	**CO_2eq Potential (Mg CO_2eq $ac^{-1}yr^{-1}$)**	**Scope**
McClelland et al. 2021[a]	0.09 (0.004)	0.31 (0.01)	Global, temperate-cool fields
Abdalla et al. 2019	0.22 (0.03)	0.80 (0.13)	Global
Poeplau & Don 2015	0.13 (0.02)	0.47 (0.06)	Global, temperate-biased
Joshi et al. 2023[b]	0.30 (0.03)	1.09 (0.12)	Global, temperate
King & Blesh 2018	0.09 (N/A)	0.31 (N/A)	Global
Jian et al 2020[c]	0.23 (N/A)	0.83 (N/A)	Global
Blanco-Canqui 2022	0.05 (N/A)[e]	0.18 (N/A)	United States
COMET-Planner	0.07 (legumes)/0.04 (non-legumes)	0.25 (legumes)/0.13 (non-legumes)	Average across all WI counties[d]

[a]If subsetting this data to temperate-cool zone, which encompasses WI, SOC potential is 0.08 (±0.01) Mg C ac^{-1} yr^{-1} or 0.30 Mg CO_2eq ac^{-1} yr^{-1} (not statistically significant from no cover crops).
[b]If subsetting to the 27 comparisons looking at 0-60 cm soil depth, the SOC potential is 0.11 (0.05) Mg C ac^{-1} yr^{-1}
[c] This study did find a negative correlation between SOC and latitude, and SOC and annual temperature
[c] For studies that found an increase, the average increase was 0.17 Mg C ac^{-1} yr^{-1}
[d] When looking at south central and southwestern counties specifically, estimated impact of cover crops is larger: 0.34 Mg CO_2e ac^{-1} yr^{-1} for legume cover crops and 0.18 Mg CO_2eq ac^{-1} yr^{-1} for non-legume cover crops

Table A.3. Alley cropping carbon sequestration rates reported in published studies, as well as COMET Planner estimates for Wisconsin						
Source	**Carbon Sequestration rate (Mg CO_2eq $ac^{-1}yr^{-1}$)**	**Location of Data**	**Species Used**	**Conversion**	**Tree Density**	**Stand Age**
Feliciano et al	2.36 (1.78 biomass; 0.58 SOC)	Biomass: global value from tropical/arid areas in Africa, Latin America and Asia. SOC: from 3 US/Canada studies	SOC: poplar and Norway Spruce No species details on biomass	Cropland to "agrisilviculture", which includes parkland, intercropping, and taungya	No density info for most studies; mix of "dense"/"sparse" when reported	Biomass: 6-50 years SOC: 6-30 years
Fargione et al.	2.15 (biomass + SOC)	Average of 6 values in published lit: 1 global review, southern France, temperate Europe, 3 Quebec/Ontario, southeastern China	Norway spruce, poplar, red oak, black cherry, white ash, walnut, orchards	Cropland to alley cropping	No density information	Not reported
Drever et al.	1.29 (1.04 biomass; 0.25 SOC)	From literature review of 8 North American studies	Hybrid poplar or "hardwood species"	Cropland to alley cropping	Standardized to 111 trees per ha	Not reported
Cardinael et al.	1.36 (1.09 biomass*; 0.28 SOC)	SOC: based on 16 studies in North America; Biomass: based on 7 studies in North America	No species information presented	Cropland to alley cropping	SOC: based on mean density of 231 trees per ha; Biomass based on mean density of 111 trees per ha	12-100, with most in the 20-40 year range
Udawatta & Jose	5.05 (biomass + SOC)	From 8 study locations in North America: GA, MO, FL, Quebec, Ontario	Mimosa/sorghum/wheat, poplar, spruce, oak, "tree-based conventional systems; pecan	Cropland to alley cropping	Not reported	1-47; average 17
COMET-Planner	1.63			Replacing 20% cropland with hardwood		NA
*If using data from all temperature/cool zones (9 studies), biomass increases to 2.08 Mg CO_2eq $ac^{-1}yr^{-1}$ with density of 271 trees per ha						

Table A.4. Windbreak carbon sequestration rates reported in published studies, as well as COMET Planner estimates for Wisconsin						
Source	**Carbon Sequestration rate (Mg CO_2eq $ac^{-1}yr^{-1}$)**	**Location of Data**	**Species Used**	**Conversion**	**Tree Density**	**Stand Age**
Feliciano et al	1.66 (biomass)	Biomass: 1 North American Study	Shelterbelt with Scots pine in prairie	Grassland to boundary planting	Not reported	40 Years
Fargione et al.	5.28 (biomass + SOC)	Average of 4 values in published lit: 3 US studies (Plains states); 1 Canadian study; 1 Chinese study	Green ash, red cedar, caragana, Siberian elm, red mulberry, cotton wood, red cedar-scotch pine, poplar, white spruce	Cropland to windbreak	Not reported	Not reported
Cardinael et al.	2.63 (1.63 biomass; 1.0 SOC)	Biomass: 12 North American studies in temperate/cool climates. SOC: 6 North American studies	Not reported	Cropland to hedgerow	Biomass: 816 trees/ha SOC:546 trees/ha	12-100, with most in the 20-40 year range
Kim et al.	2.08 (1.63 biomass; 0.45 SOC)	2 studies (US, Canada)	Red cedar, Scotch pine	grassland to shelterbelt	Not reported	Not reported
Udawatta & Jose	1.43 (biomass + SOC)	One North American study	Hybrid poplar and white spruce	Cropland to windbreak	40 trees/ha	35
COMET-Planner	2.97-5.94			Replacing strip of cropland with conifer/hardwood		NA

Table A.5. Silvopasture carbon sequestration rates reported in published studies						
Source	**Carbon Sequestration rate (Mg CO_2eq $ac^{-1}yr^{-1}$)**	**Location of Data**	**Species Used**	**Conversion**	**Tree Density**	**Stand Age**
Feliciano et al	2.36 (1.78 biomass; 0.58 SOC)	Biomass: 1 US study; SOC 3 US/Canada studies	douglas fir/ryegrass/clover; poplar on grassland	Grassland to silvopasture	Not reported	Biomass; 11 years SOC: 11-18 years
Drever et al.	1.23 (0.94 biomass; 0.30 SOC)	Determined from review of 5 studies from North America	Deciduous trees	Biomass and carbon accumulation following the introduction of trees into existing pasture	Not reported	Not reported
Cardinael et al.	2.06 (1.57 biomass*; 0.28 SOC)	Biomass: 1 NA cool/temperate study; SOC: 6 temperate/cool studies (5 Europe, 1 South America)	Not reported	Grassland to silvopasture	Biomass: 283 trees/ha SOC:546 trees/ha	12-100, with most in the 20-40 year range
Kim et al.	5.64 (2.67 biomass; 2.97 SOC)	13 global studies (1 US study, majority from India)		Grassland to silvopasture	Not reported	5-11 years, when reported
Udawatta & Jose	9.06 (biomass + SOC)	4 US study locations (OR, FL)	Douglas fir/ryegrass/clover; pine/bahia grass	Grassland to silvopasture	Not reported	Not reported
*All cool/temperate regions: 6 studies with average tree density of 312 trees: total biomass 3.4 Mg CO_2eq $ac^{-1}yr^{-1}$						

Source	Carbon Sequestration rate (Mg CO_2eq $ac^{-1}yr^{-1}$)	Location of Data	Species Used	Conversion	Tree Density	Stand Age
Feliciano et al	5.24 (5.54 biomass; -0.30 SOC)	Biomass: 1 Canadian study; SOC: global; largely tropical	Biomass: poplar /green ash on prairie SOC: largely eucalyptus/acacia	Biomass: grasslands to woodlots; SOC: Cropland/fallow to woodlot	Biomass: “dense” trees	SOC: Mostly 4-5 years, but one 31 and one 50; Biomass: 40 years
Drever et al.	Biomass: 0.11*stand age (year) +0.30; SOC: 1.33 (Corresponds to 2.68 for a 10 year stand, 3.74 for a 20 year stand)	Biomass: Determined from review of 3 biomass stock studies from North America; SOC: average from 4 studies of SOC accumulation following forest cover restoration in North America	Deciduous trees	Reflective of growth potential in productive riparian sites of southern Ontario	Not reported	Not reported
Kim et al.	1.19 (all biomass)	1 US (IA) location (series of 3 studies)	Poplar/switchgrass	Cropland to riparian buffer	Not reported	Not reported
Udawatta & Jose	3.86 (biomass + SOC)	11 study locations in: IA, NY, SC, Ontario, WA	Poplar/switchgrass, mixed hardwoods in natural riparian systems	Cropland/grassland to riparian buffer	Not reported	Biomass 1-250 years; SOC: 2-60 years
COMET-Planner	5.94-7.42			Replace strip of cropland/grassland with mixed hardwoods		NA

Table A.6. Riparian Buffer carbon sequestration rates reported in published studies, as well as COMET Planner estimates for Wisconsin

Table A.7. Summary of soil carbon sequestration potential of conversion from annual crops to perennial herbaceous crops as reported in published reviews and meta-analyses.				
Source	**Soil carbon sequestration (Mg CO_2eq $ac^{-1}yr^{-1}$)**	**Perennial Crop**	**Geographic Scope**	**Soil Depth Sampled**
Ledo et al. 2020	0.40	Switchgrass and *Miscanthus*	Global temperate regions, at least 10 years since conversion	50-100 cm
King & Blesh 2018	0.61	Mostly alfalfa, but some legume/grass mixtures	Global, but heavy North America bias (63% of sites included); median time since conversion is 14 years.	20 cm
Angostini et al. 2015	1.69-2.79	Switchgrass, *Miscanthus*	Not reported; generally short-term (<6 years) studies	150 cm
Qin et al. 2016	1.62 (*Miscanthus);* 1.90 (switchgrass)	Switchgrass, *Miscanthus*	Global; North America and Europe bias	100 cm

Table A.8. Summary of direct N_2O emissions factor (N_2O-N emissions from nitrogen fertilizer as a percentage of total N-input) or equations describing N_2O-N emissions as a function of N-input calculated from bottom-up approaches.					
Source	**Global**	**US**	**US Corn**	**US North Central Region**	**Notes**
IPCC 2019	1% (0.2-1.8%)				Synthetic N fertilizer in wet climate: 1.6% (1.3-1.7%)
Grace et al. 2011				1.75%	
Griffis et al. 2013				1.3%	
Hoben et al. 2010				[4.36 + 0.025 N]xN (0.56-0.93% for 50-200 kg N input/ha)	Michigan experimental plots; eqn provides g N_2O-N per ha and requires kg N input per ha.
Shcherbak et al. 2014	[6.49 + 0.0187N]xN (0.74-1.02% for 50-200 kg N input /ha)				Global meta-analysis; this eqn is for upland grain crops; requires kg N input per ha and outputs g N_2O-N per ha.
Gerber et al. 2016	0.77%	0.83%	0.92%		Global meta-analysis; See Table S12 for overall model calculations for different levels of N_2O emissions per N added with 95% CIs.

Table A.9. Summary of top-down estimates of N_2O emissions factors for nitrogen fertilizer reported in published literature		
Source	**EF**	**Scope**
Mosier et al. 1998	5.5%	Global
Prather et al. 2001	2.6-5.5%	Global
Crutzen et al. 2008	3-5%	Global
Davidson 2009	2.5% for fertilizer N and 2% for manure N.	Global
Griffis et al. 2017	5.3%	US Corn Belt
Thompson et al. 2019	2.3 (±0.6)%	Global

Table A.10. Comparison of N_2O emissions factors from nitrogen fertilizer in the US Environmental Protection Agency's state inventory tool (used in the Wisconsin Department of Natural Resources Greenhouse Gas Inventory report) and factors from the most recent Intergovernmental Panel on Climate Change report

Parameter	**EPA SIT/WDNR GHG Inventory**	**IPCC 2019 for wet cool climates**
Direct EF (%)	1	1.6
Synthetic N volatilized (%)	10	11
Synthetic N leach/runoff (%)	30	24
Indirect EF (%)	1	1
Leach/Runoff EF (%)	0.75	1.1
Total EF (kg N_2O-N/kg N fert)	1.20	1.86

Table A.11. Methane conversion factors and N_2O emissions factors for various manure management practices from the IPCC and EPA

Manure Management Practice	Methane Conversion Factor (%)	N_2O Emission Factor (%)
Pasture	0.5	0[a]
Daily Spread	0.1 (cool moist climate)-0.5 (temperate moist climate)	0[a]
Solid Storage	2 (cool moist climate)-4 (temperate moist climate)	1
Deep Pit	24.1[b]	0.2
Liquid/Slurry	24.1[b]	0.5
Anaerobic Lagoon	67.5 (uncovered)[b]	0 (uncovered) - 0.0.5 (covered)
Anaerobic Digester	2.9[c] (1-10)[d]	0.06

[a]N_2O emissions associated with pasture-deposited manure and daily spread are accounted for under emissions from managed soils. For the purposes of manure management emissions category, N_2O emissions are considered zero

[b]Calculated for Wisconsin's climate by the EPA using the van't Hoff-Arrehenius equation recommended by the IPCC.

[c]EPA estimate for anaerobic digester systems in Wisconsin

[d]Range of EFs from IPCC.

Table A.12. Estimated percent of manure handled by different practices in 2018 by the EPA, along with each practice's methane conversion factor (MCF). The overall state weighted MCF is calculated by summing the product of each practice's proportion of manure handled by its MCF.		
Manure Management Strategy	**Percent of Manure Handled in WI**	**Methane Conversion Factor (%)**
Pasture	14.9	1
Daily Spread	5.4	0.1
Solid Storage	24.2	2
Liquid/Slurry	3.2	24.1
Deep Pit	22.7	24.1
Anaerobic Lagoon	23.7	67.5
Anaerobic Digester	5.9	2.9

Table A.13. Carbon intensity of grazed vs confined milk production in Wisconsin from Reinemann & Cabrera 2013.		
Greenhouse gas emissions (kg CO_2eq per kg FPCM)	**Grazed**	**Confined**
Enteric emissions	0.37-0.44	0.39
Manure	0.12-0.15	0.18
On-farm energy use	0.04-0.05	0.06
Crops and feeds on-farm	0.03-0.07	0.1
Crops and feeds off-farm, other off-farm inputs	0.03-0.04	0.04
Total	**0.6-0.75**	**0.77**

Table A.14. Carbon intensity of grazed vs confined milk production in Wisconsin from Cabrera & Dutreuil 2014.		
Greenhouse gas emissions (kg CO_2eq per kg FPCM)	**Grazed**	**Confined**
Enteric + barn CH_4	0.34-0.49	0.3
Manure storage	0	0.09
On farm fuel combustion	0.019-0.028	0.023
Feed production (includes emissions from field applied manure)	0.13-0.14	0.08
Secondary sources (manufacture of fuel, machinery, fertilizers, pesticides, etc.)	0.04-0.06	0.1
Total	**0.48-0.7**	**0.58**

Table A.15. Carbon intensity of grazed vs confined milk production in Wisconsin from Dutreuil et al. 2014.		
GHG Emissions (kg CO_2eq per kg ECM)	**Grazed**	**Confined**
Enteric + barn CH_4	0.45-0.68	0.45
Manure storage	0-0.05	0.132
On farm fuel combustion	0.02-0.04	0.035
Feed production	0.12-0.18	0.114
Secondary sources	0.08-0.16	0.149
Net biogenic (includes soil carbon sequestration)	(0.28)-(0.31)	(0.3)
Total, not including biogenic	**0.76-1.01**	**0.88**

Table A.16. Carbon intensity of grazed vs confined milk production in Wisconsin from Aguirre-Villegas et al. 2017		
GHG Emissions (kg CO_2eq per kg FPCM)	**Grazed**	**Confined**
Enteric CH_4	0.47-0.51	0.49
Manure (barn + storage)	0.08-0.09	0.1
On farm energy use	0.18-0.19	0.14
Crop production (incl. land application of manure)	0.06-0.08	0.08
Off-farm emissions	0.05-0.05	0.06
Total	**0.85-0.92**	**0.87**

Table A.17. Carbon intensity of grazed vs confined milk production in Wisconsin from CIAS 2019		
GHG Emissions (kg CO_2eq per kg FPCM)	**Grazed**	**Confined**
Enteric CH_4	0.47-0.51	0.48
Manure (barn + storage)	0.08-0.09	0.23
On farm energy use	0.18-0.2	0.139
Crop production	0.06-0.07	0.078
Off-farm emissions	0.06	0.064
Total	**0.85-0.93**	**0.991**

Table A.18. Carbon intensity of grazed vs confined milk production in Wisconsin from Aguirre-Villegas et al. 2022		
GHG Emissions (kg CO_2eq per kg FPCM)	**Grazed (organic)**	**Confined**
Enteric CH_4	0.66	0.49
Manure CH_4+N_2O	0.23	0.24
Energy	0.17	0.12
Soils N_2O	0.16	0.09
Inputs	0.05	0.04
Total	**1.27**	**0.98**